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# *ARGULUS YUCATANUS* N. SP. (CRUSTACEA: BRANCHIURA) PARASITIC ON *CICHLASOMA UROPHTHALMUS* FROM YUCATAN, MEXICO

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*ABSTRACT* A new species, *Argulus yucatanus*, is described based on 14 specimens from *Cichlasoma urophthalmus* collected in Celestun Lagoon, Yucatan, Mexico. Diagnostic characters include the number of and shape of sclerites in the suction cup support rods, shape of and position of respiratory areas, and modifications on the legs of males. In males, the coxae of the 2nd legs bear an angular lobe with 5–7 erect scales and 13–21 sensilla. The new species is compared to *Argulus funduli* Krøyer, 1863, *A. chromidis* Krøyer, 1863, *A. cubensis* Wilson, 1936, *A. rhamdiae* Wilson, 1936, and *A. varians* Bere, 1936.

*RESUMEN* Una nueva especie, *Argulus yucatanus*, está escrito de catorce especimenes de *Cichlasoma urophthal-mus* colectaron del Estero de Celestún, Yucatán, México. Varios caracteres la distinguen con inclusión del numeró de y de la forma de escleritos en las rayas de las ventosas, de las áreas respiratorias y de las modificaciónes en las patas de los machos. En los machos, las segundas parejas de las patas tienen un lóbulo angular con 5–7 escamas erguidas y 13–21 sensillas. *Argulus funduli* Krøyer, 1863, *A. chromidis* Krøyer, 1863, *A. cubensis* Wilson, 1936, *A. rhamdiae* Wilson, 1936 y *A. varians* Bere, 1936 están comparado a la nueva especie.

## INTRODUCTION

Only 3 species of *Argulus* have been described from Mexico (Wilson 1936a, Pineda et al. 1995, Poly 2003), namely *A. rhamdiae* Wilson, 1936, *A. mexicanus* Pineda, Páramo and Del Rio, 1995, and *A. ambystoma* Poly, 2003. In addition, 4 other species of *Argulus* have been listed as components of the Mexican fauna (see Poly 2003). The present study includes a description of one new species from Mexican waters and comparisons of the new species with other species that are either similar to it in some features or that occur in the region. Also, new data and illustrations are included for 2 poorly known species, *A. chromidis* Krøyer, 1863 and *A. cubensis* Wilson, 1936.

## MATERIALS AND METHODS

All specimens were fixed and stored in 4% formalin in 1994 and were transferred to 70% ethanol in 1998. Six females and 8 males (13 mature, 1 immature male) were examined under dissecting and compound microscopes in a watchglass or as a temporary slide mount (with 70% ethanol and Hoyer's medium). Drawings were made with the aid of a camera lucida. All measurements were made using an ocular micrometer, and measurements reported below are arranged as follows: range (mean, holotype) with allotype values substituted for females. Width of first antennae refers to the distance from the mesial margin of the basal segment to the farthest extent of bend in the terminal spine on the 2nd segment. Two males and one female also were examined using scanning electron microscopy, and preparation procedures were modified slightly from those of Rupp (1990). Specimens were dehydrated in an ethanol series consisting of 80% (5 min), 90% (5 min), 100% (1st, 5 min; 2nd, 10 min), then critical point dried, mounted on metal stubs with carbon paint, allowed to dry in an oven (60 °C), and sputter coated with gold/palladium. Type specimens were deposited in the American Museum of Natural History, New York (AMNH), in the Museum of Comparative Zoology, Cambridge, Massachusetts (MCZ), and in the author's collection. The syntypes of *A. funduli* (ZMUC CRU-6473, 4 males, 2 females [data collected from 2 males and 2 females]), the holotype of *A. chromidis* (ZMUC CRU-6030, 1 female, poor condition) (both from the Zoologisk Museum, Copenhagen, Denmark), and the syntypes, along with other specimens, of *A. cubensis* (MCZ 8973 [syntypes], 1 male, 1 female; MCZ 9643 [non-types], 2 females) were examined for comparison with the new species using the methods described above. Information about *A. varians* was obtained from literature sources (Bere 1936, Bouchet 1985).

## RESULTS

### Family Argulidae Rafinesque, 1815
### *Argulus* Müller, 1785
### *Argulus yucatanus*, n. sp.
### Yucatan fishlouse
### Figures 1–7, Tables 1–2

**Material examined.** Holotype—adult male, 2.83 mm total length, AMNH Crustacea 18469, Estero de Celestún (Celestun Lagoon), Yucatán, México, 4 November 1994,

collector: Frantiπek Moravec. Allotype: adult female, 3.33 mm total length, AMNH Crustacea 18470. Other paratypes: 1 adult male, 1 adult female, AMNH Crustacea 18471; 2 adult males, 1 immature male, 2 adult females, MCZ 50725; 3 adult males, 2 adult females (includes SEM specimens), author's collection; all paratypes collected with holotype.

**Diagnosis.** Two respiratory areas with smaller, circular to ovoid "area" anterior to larger posterior "area;" postantennal spines single; males with 37–46 support rods per suction cup; females with 43–51 support rods per suction cup; usually 2–3 sclerites per rod in males, 3–5 sclerites per rod in females; mouth tube with 2–5 (usually 4) scales on basal half; bushy fringe of fine setae on margin of labium; basal plate of 2nd maxilla with 3 stout, digitate spines, larger scales on anterior of pad with smaller scales posteriorly, scales round to ovoid with coarse-pectinate margin, 3–8 stout, naked setae on posterior margin of pad; first 2 pairs of legs lacking flagella; posteroventral surface of coxae of 2nd legs of males with an angular, fleshy lobe bearing 5–7 erect scales on posterior margin and 13–21 sensilla; 3rd legs of males with 2 ornamented pegs issuing from cleft on dorsal side of legs; male abdomen much longer and narrower than female abdomen; spermathecae of female ovate, relatively large; testes of male extend entire length of abdomen; caudal rami long, slender, basal in anal sinus.

**Description.** Total length (mm) 2.17–3.38 (2.88, 2.83) in males, 2.80–3.88 (3.40, 3.33) in females. Carapace shape as shown in Figures 1A, B, with cephalic region distinctly separated from alae. Carapace length (mean of both alae, mm) 1.30–2.00 (1.73, 1.69) in males, 1.95–2.67 (2.42, 2.39) in females. Maximum carapace width (mm) 1.18–2.00 (1.68, 1.65) in males, 1.75–2.55 (2.21, 2.12) in females. Carapace extending as far as anterior margin of to middle of 3rd legs in males, as far as anterior margin of 3rd legs to anterior margin of 4th legs in females. Females with eggs in thorax but not in carapace alae. Sensilla and pores scattered on dorsal surface and margins of carapace; fringe of small sensilla with larger sensilla interspersed along cephalic margin of carapace. Pair of compound eyes anteriorly with diameters (left and right eyes, μm) 100–150 (122, left: 110, right: 110) in males, 120–150 (130, left: 130, right: 130) in females. Transverse distance between eyes (μm) 340–510 (421, 420) in males, 480–630 (558, 520) in females. Nauplius eye with one anterior and 2 posterior ocelli. Sclerotized dorsal ridges not forked anterior of eyes. Ventrally, carapace with small, posteriorly-projecting spines along outer margin, more numerous anterior of respiratory areas with few spines occurring beyond anterior margin of larger res-

piratory area. Respiratory areas consist of smaller, circular to ovoid "area" anterior to larger, posterior ovoid "area" (Figures 1C–E); respiratory areas not outlined with pigment (possibly lost in preservative). Color in preservative white to light yellow; no other pigmentation present (possibly lost in preservative).

Thorax compressed dorsoventrally, 4-segmented, with 2 pairs of posteriorly-projecting spines ventrally. Spines digitate, anterior pair (accessory spines) usually larger than posterior pair (postmaxillary spines). Accessory spines between basal segments of 2nd maxillae. Postmaxillary spines farther apart than accessory spines. Males with ovoid fleshy lobe at posterior of 4th thoracic segment between natatory lobes (Figures 2A, B). Thorax with coarse-pectinate scales scattered on ventral surface (Figure 2A). Dorsal surface of thorax with sensilla; typically one sensillum at midline of posteror margins of 2nd and 3rd thoracic segments with others variously placed (Figure 2C). Four pairs of biramous swimming legs composed of a precoxa, coxa, basis, exopod, and endopod (Figure 2A). Exopods and endopods with plumose setae. Setae usually absent from coxae of 2nd and 3rd legs; see Table 1 for number of setae on legs. First 2 pairs of legs lacking flagella. Endopods of first pair of legs 3-segmented with 3 setae distally. Endopods of 2nd pair of legs unsegmented. Endopods of 3rd and 4th pairs of legs 2-segmented. Second, 3rd, and 4th legs of males with secondary sexual structures (Figure 2A). Coxae of male 2nd legs with an angular, fleshy lobe posteroventrally with 5–7 (6, 7) erect scales on posterior margin and 13–21 (17, 19–21) sensilla (Figure 2D); number of erect scales on angular lobe excludes prostrate coarse-pectinate scales present on ventral surface of coxae. Dorsal surface of coxae of male 3rd legs covered with closely-arranged fine-pectinate scales (Figures 1A, 2E). Two ornamented pegs issue dorsally from joint between coxa and basis of male 3rd legs with fine-pectinate feather-like scales posterior and ventral to the pegs (Figures 2E, F, 3A, B). Pegs with horn dorsally; orifice of pegs containing many bi-pronged and multi-pronged projections from inner wall. Dorsal surface of precoxae of male 3rd legs with many small sensilla. Bases of male 4th legs with 2 opposing blunt lobes on anterior surface; dorsal lobe with scaled area anteriorly (non-pectinate scales) and small patch of tubercles distally (Figures 3C, D). Precoxae and coxae of male and female 4th legs with posterior natatory lobes fringed with plumose setae and bearing scales and sensilla. Female natatory lobes with more scales and sensilla than those of male. Bases of 4th legs of both sexes larger than bases of other legs. Coarse-pectinate scales on ventral surfaces of precoxae and coxae (Figure 3E).
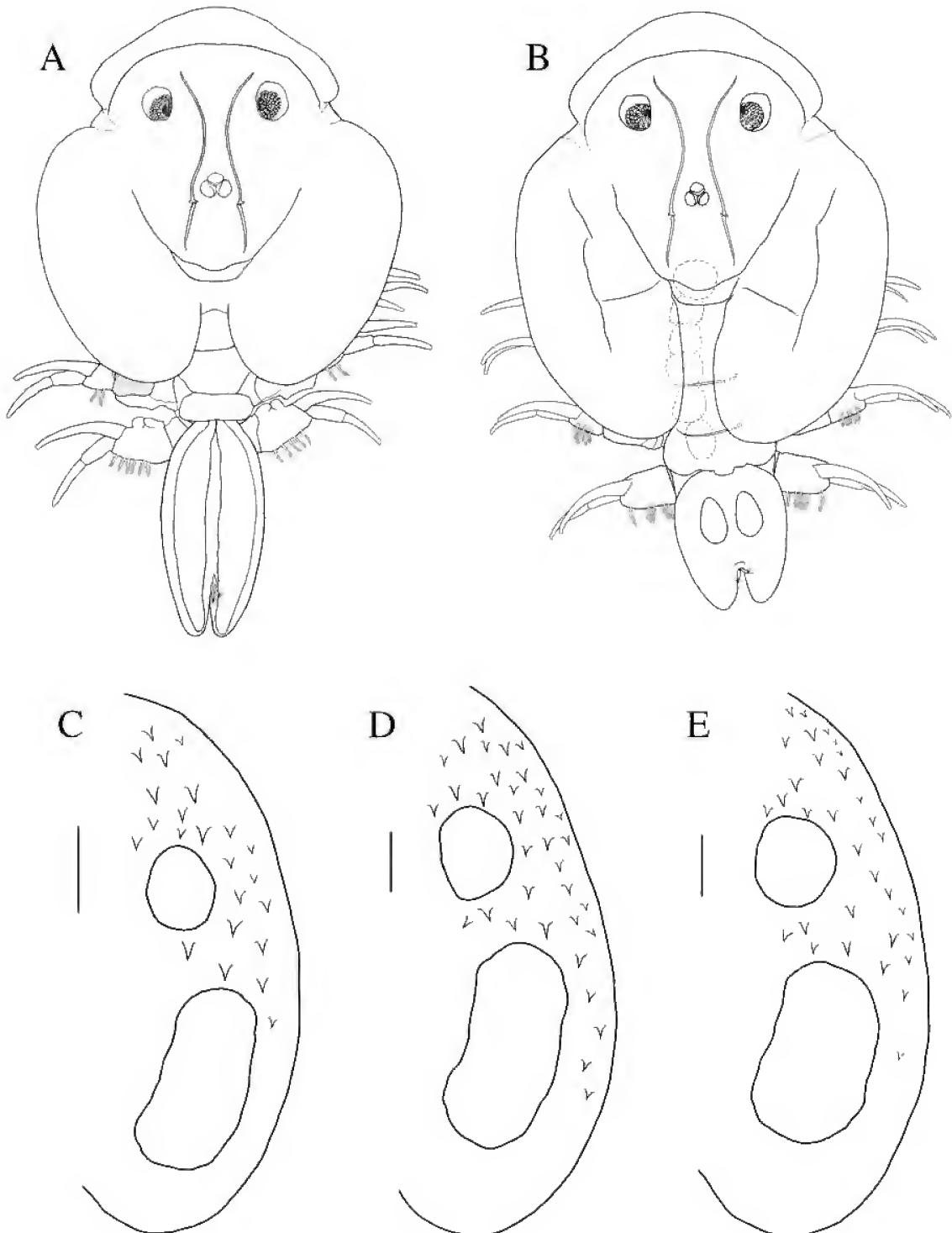
**Figure 1.** *Argulus yucatanus*, **n. sp. A) Male, dorsal view (holotype, 2.83 mm total length, AMNH Crustacea 18469). B) Female, dorsal view (allotype, 3.33 mm total length, AMNH Crustacea 18470). C–E) Shape of respiratory areas and distribution of adjacent spines on: C) holotype (male), D) allotype (female), E) paratype (female, AMNH Crustacea 18471). For clarity, plumose setae were not shown on endopods and exopods of legs (1A, B). Number of setae illustrated on bases is actual number present on these particular specimens (1A, B); dorsal and ventral rows of setae can be seen on the bases of 3rd legs (1B). Eggs in thorax shown by dashed lines (1B). Scale: C–E = 200 µm.**
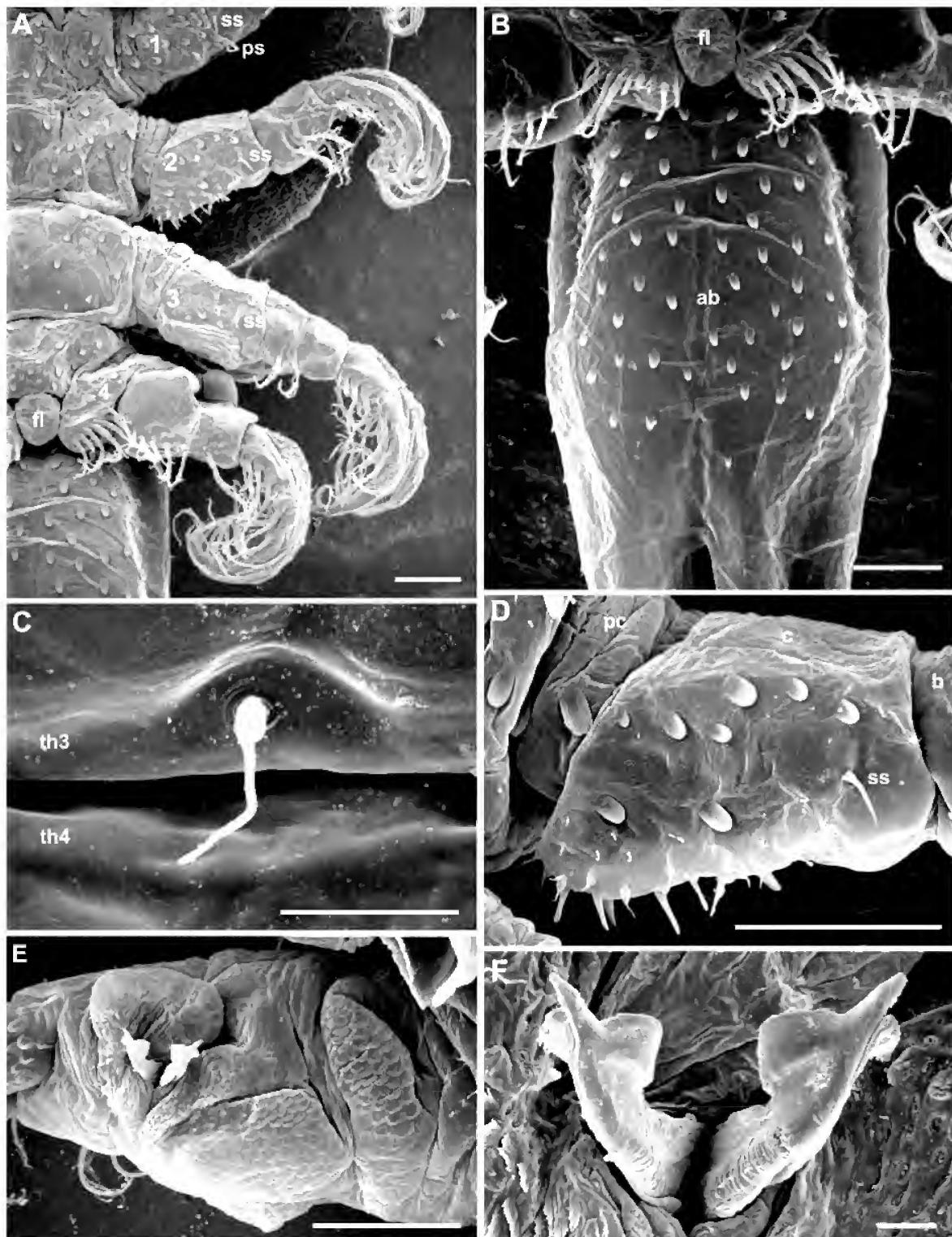
Figure 2. *Argulus yucatanus*, n. sp. (males). A) Ventral view of legs, thorax, and abdomen; legs numbered 1–4 on coxae, ss = simple seta, ps = plumose seta, fl = fleshy lobe. B) Distribution of scales on ventral surface of abdomen (ab) and fleshy lobe (fl) between natatory lobes. C) Sensillum at midline of thorax at posterior margin of 3rd thoracic segment (th3), extending over 4th thoracic segment (th4). D) Angular, fleshy lobe on coxa of 2nd leg (ventral view); pc = precoxa, c = coxa, b = basis, ss = simple seta. E) Pair of pegs and fine-pectinate scales covering portions of coxa of 3rd leg (dorsal view). F) Close-up of pegs on 3rd leg (dorsal view). Scale: A, B, D, E = 100 µm; C, F = 10 µm.

**TABLE 1**

**Number of setae on coxae and bases of legs of *Argulus yucatanus*, n. sp. Separate counts of both right and left legs for each individual were included (8 males, 6 females; range followed by mean in parentheses). Most setae were plumose; some setae have small plumes and sometimes appeared simple. There is usually a single, simple seta or sensillum not associated with the others on the coxae and bases of legs 1–3, and these were not included in the counts.**

|  |  | Coxa (Ventral) | Basis (Ventral) | Basis (Dorsal) |
|---|---|---|---|---|
| Leg 1 | Male | 1–2 (1) | 2–4 (3) | 0 |
|  | Female | 1 (1) | 2–3 (3) | 0 |
| Leg 2 | Male | 0–1 (0)[a] | 1–3 (3) | 1–4 (3) |
|  | Female | 0 | 2–4 (3) | 3–5 (4) |
| Leg 3 | Male | 0 | 1–2 (2) | 0 |
|  | Female | 0 | 2–4 (3) | 3–4 (4) |
| Leg 4 | Male | 6–10 (8) | 4–6 (5) | 0 |
|  | Female | 13–23 (20) | 6–10 (8) | 0 |

[a]Only one of 16 legs from 8 specimens with a seta; setae usually not present on segment

Abdomen bilobate. Each lobe with single row of small, coarse-pectinate scales along posterolateral edges and small sensilla near tips and along lateral margins in both sexes. Male abdomen longer and narrower than female abdomen (Figures 1A, B). Abdomen length (mm) 0.76–1.09 (0.99, 1.00) in males, 0.68–0.93 (0.83, 0.82) in females; maximum width (mm) 0.45–0.55 (0.50, 0.49) in males, 0.46–0.71 (0.64, 0.65) in females. Anal sinus length (μm) 200–320 (282, 290) in males, 290–340 (318, 320) in females. Caudal rami paired, long, slender, at base of anal sinus; each ramus with 5 stout, naked "setae" (Figure 3F). Spermathecae of female paired, brownish, ovate, relatively large, located anteriorly on abdomen (Figure 1B). Abdominal papillae absent on female abdomen. Testes of male occupy much of abdomen, extending entire length of abdominal lobes (Figure 1A). Abdomen of male with coarse-pectinate scales on ventral surface anterior of anal sinus (Figure 2B).

First antennae 4-segmented. First segment (basal segment) sclerotized, large with stout posteriorly-projecting posterior spine; 2nd segment sclerotized with small recurved spine anteriorly, posteriorly-projecting medial spine, and large recurved terminal spine; 3rd segment fleshy, cylindrical with large, stout seta distally that projects ventrally and several smaller setae; 4th segment fleshy, small, with few setae distally (Figures 4A, B). Width of first antennae (mean of both antennae, μm) 210–280 (256, 255) in males, 280–360 (330, 325) in females. Second antennae 5-segmented, fleshy. First 2 segments larger; remaining 3 thin, cylindrical; basal segment bears posteriorly-projecting posterior spine. All segments of 2nd antennae with several long, stout setae that project distally; some

reaching to or beyond junction with next segment. Postantennal spines single (as opposed to double in some taxa), large, rounded or pointed distally (Figures 4A, B).

First maxillae modified into suction cups in adults. In males, first maxillae inner diameter (μm) 220–290 (241, left: 230, right: 220) (*n* = 14, left and right) and outer diameter (μm) 310–420 (352, left: 350, right: 340) (*n* = 14). In females, inner diameter (μm) 370–450 (428, left: 410, right: 410) (*n* = 12, left and right) and outer diameter (μm) 500–650 (595, left: 570, right: 570) (*n* = 12). See Table 2 for number of support rods in suction cups. Number of sclerites per support rod in males 1–4 (2, 2, range 1–3) (*n* = 596 support rods) and in females 1–5 (4, 4, range 1–5) (*n* = 549 support rods). Usually 2–3 sclerites per rod in males, 3–5 sclerites per rod in females; lower counts such as one due to missing sclerite(s) or atypical development, uncommon. Number of sclerites variable with position on suction cup; shape of sclerites variable; orientation of rods changes at 2 points on rim of suction cup (Figure 5, see Discussion). Basal (proximal) sclerite usually rod-shaped, longer than other sclerites. Distal sclerites bowl- or cylinder-shaped. Suction cups with 9–15 (12, 12) sensilla on inside circumference; sensilla with pore at tip and with or without tentacles distally (Figures 6A–C). Short, conical sensilla, with pore distally, on rim of suction cup between basal sclerites of some support rods (Figures 6A, D).

Second maxillae 5-segmented with broad basal plate bearing 3 stout, digitate spines, usually larger space between lateral spine and central spine (Figures 6E, F). Basal plate with elevated pad bearing 3–4 large (anteriorly) and more smaller coarse-pectinate scales and 3–8 stout setae that

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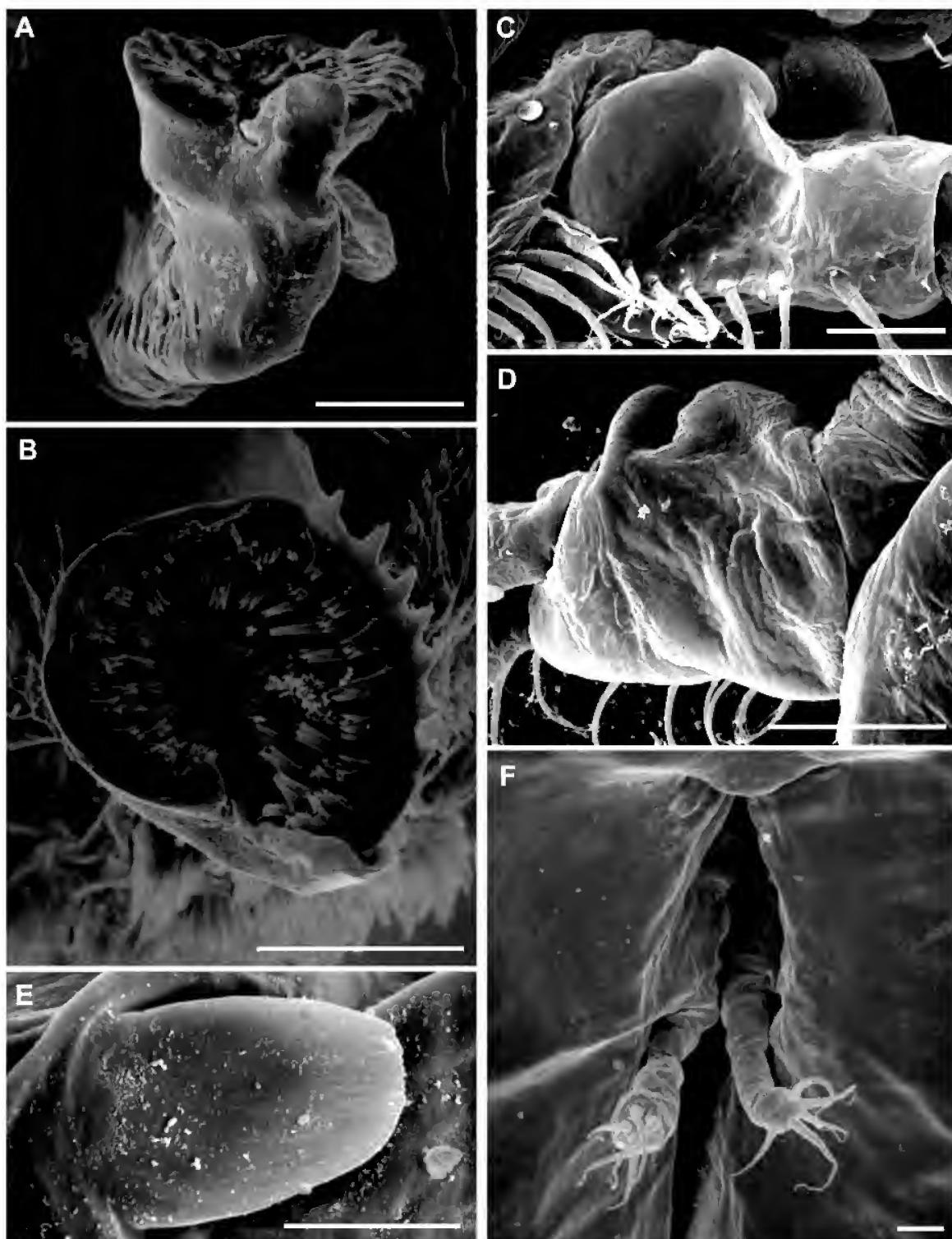
**Figure 3.** *Argulus yucatanus*, n. sp. (males). A) Peg on 3rd leg (dorsal view), rotated partially toward anterior face. B) Anterior face of a peg illustrating detail of ornamentation; note feather-like scales below the peg. C) Ventral view of basis of 4th leg with 2 opposing lobes, one of which has a scaled area anteriorly (non-pectinate scales) and patch of small tubercles distally. D) Dorsal view of basis of 4th leg. E) Coarse-pectinate scale on base of mouth tube; this is the typical coarse-pectinate scale found on ventral side of thorax, mouth tube, coxae, and basal plate of 2nd maxillae of both sexes and ventral side of abdomen of males. F) Caudal rami at base of anal sinus of abdomen (dorsal view, male). Scale: A, B, E, F = 10 μm; C = 50 μm; D = 100 μm.

**Figure 4.** *Argulus yucatanus*, n. sp. A, B) First and 2nd antennae and postantennal spine (ventral view; A, male; B, female). Scale: A, B = 100 µm.

extend posteriorly, usually over space between central and lateral posterior spines (Figures 6E, F). Scales and setae/sensilla on ventral surfaces of last 4 segments. Distal segment with 2 sharp claws and 1 blunt, elongate lobe positioned above claws, with small sensillum at tip of lobe.

Mouth tube of moderate length, usually not reaching thoracic accessory spines, with 2–5 (usually 4) scales on basal half (Figures 3E, 7A). Labium with fringe of fine setae around mouth (Figures 7A, B); labrum with embedded scales around mouth (Figure 7C). Two pairs of sensilla on both labium and labrum; 3 other pairs of similar structures on mouth tube (Figure 7B). Pair of serrated mandibles and pair of labial ducts inside mouth tube (Figure 7C). Preoral stylet present, when extended usually reaching as far as area anterior of first antennae nearly to anterior margin of carapace (Figure 4C). Short projections and orifice near tip of stylet (Figure 7D).

**Host.** *Cichlasoma urophthalmus* (Günther), Mayan cichlid.

**Etymology.** The specific name, *yucatanus*, is derived from the state in which the type locality is located (Yucatan, Mexico) and is treated herein as a noun in apposition.

**Remarks.** Male *A. yucatanus* and *A. funduli* Krøyer, 1863 are quite similar in the shapes of the cephalic region, carapace, and abdomen as seen from a dorsal view; however, *A. yucatanus* can be distinguished from *A. funduli* by the much lower number of sclerites in the suction cup support rods (2–5 vs. 11–26, respectively ($n = 4$, *A. funduli*, mean = 17; 290 rods)), and *A. yucatanus* also has fewer support rods per suction cup than *A. funduli* (37–51 vs. 53–64, respectively ($n = 4$, *A. funduli*, mean = 58; 8 suction cups)). *Argulus yucatanus* has a pair of accessory spines and a pair of postmaxillary spines, whereas *A. funduli* lacks both pairs of spines. *Argulus funduli* has more

**TABLE 2**

**Number of support rods in first maxillae (suction cups) of male and female *Argulus yucatanus*, n. sp. (numbers for holotype and allotype in bold). [a]Not counted**

| | Male (*n* = 8) | | | | | | | | Female (*n* = 6) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Left Suction Cup | **37** | 42 | 41 | 44 | 45 | 42 | 46 | 40 | **51** | 46 | 43 | 49 | 48 | 44 |
| Right Suction Cup | **42** | 44 | 43 | 43 | 43 | 43 | 42 | —[a] | **48** | 48 | 46 | 47 | 47 | 46 |
| | $\bar{x} = 43$; Range = 37–46 | | | | | | | | $\bar{x} = 47$; Range = 43–51 | | | | | |
| | | | | | $\bar{x} = 44$; Range = 37–51 (male and female) | | | | | | | | | |
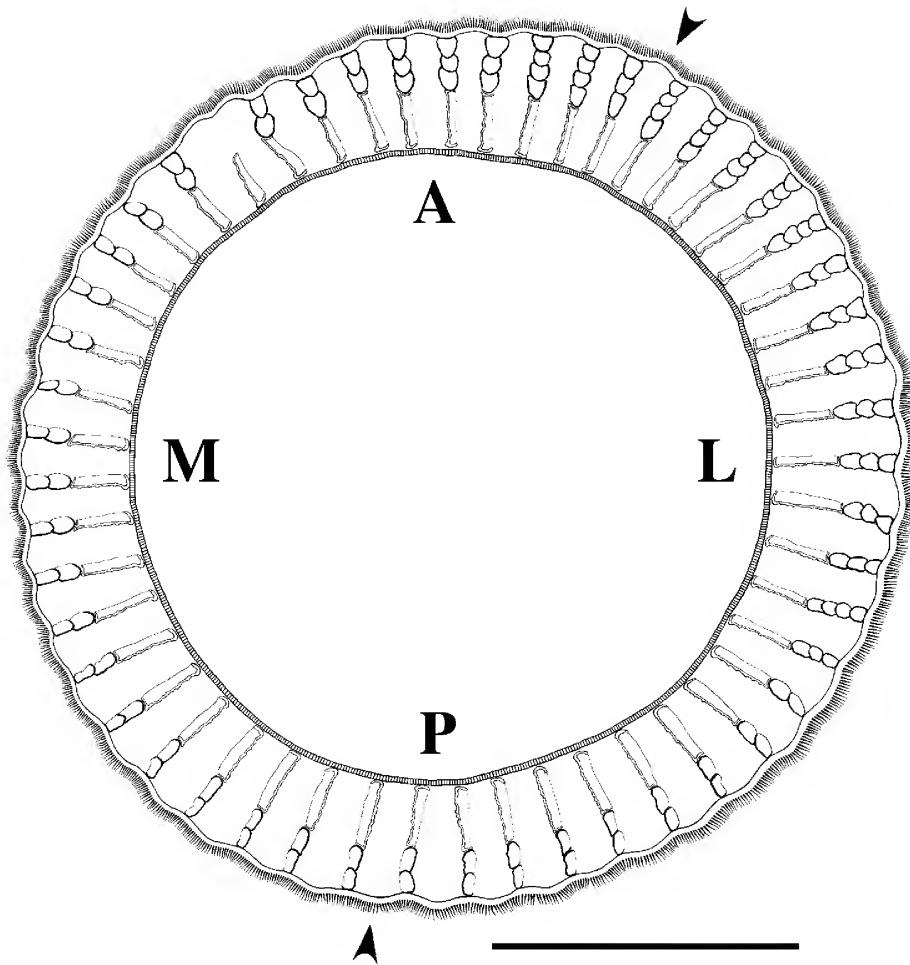
**Figure. 5.** *Argulus yucatanus*, n. sp. Suction cup (first maxilla) rim, support rods, and fringe of setae (allotype, AMNH Crustacea 18470, left side); arrows indicate points at which orientation of rods changes; A = anterior, P = posterior, L = lateral, M = medial. Note missing sclerites in one rod. Scale: 200 μm.

scales on the mouth tube than *A. yucatanus*. Male *A. funduli* do not have the secondary sexual modification on the coxae of the 2nd legs as do male *A. yucatanus*. The shape and position of the respiratory areas are similar, but not identical, between the 2 species, and both differ in features of the basal plate of the 2nd maxilla.

*Argulus varians* Bere, 1936 resembles *A. yucatanus* in shapes of the cephalic region and carapace, absence of flagella on legs, and number and shape of sclerites in suction cup rods, but they can be distinguished from each other by the size of and shape of the respiratory areas and differences in the shape of the abdomen. The natatory lobes of female *A. varians* possess a projection posterolaterally that is not present on the natatory lobes of female *A. yucatanus*, and the spermathecae of *A. varians* differ from those of *A. yucatanus*, following information given in Bere (1936) and Bouchet (1985). Male *A. varians* differ from male *A. yucatanus* in the secondary sexual modifications on the legs.

*Argulus chromidis* and *A. cubensis* both have flagella on the first 2 pairs of legs, round spermathecae, and eggs in the carapace alae of gravid females (in addition to those in the thorax); however, Krøyer (1863) exaggerated the egg distribution in the carapace of *A. chromidis* in his drawing (the hexagonal pattern on the specimen). In addition, the number of setae on leg segments, shape of respiratory areas, shape of body, and features of the 2nd maxillae of both species differ markedly from *A. yucatanus* (Figures 8B, C, E, F). Secondary sexual modifications on the legs of male *A. cubensis* differ from those of male *A. yucatanus*. *Argulus cubensis* has 39–49 support rods (*n* = 4, *A. cubensis*, mean = 45; 7 suction cups) and 4–6 sclerites per rod (mean 4, 217 rods; Figure 8D). *Argulus chromidis* lacks armature on the mouth tube, and the single suction cup available for *A. chromidis* had 42 support rods and 3–5 sclerites per rod (mean = 4; 41 rods; Figure 8A). The brief description of *Argulus rhamdiae* (based on a single female) does not agree with *A. yucatanus* in body shape,
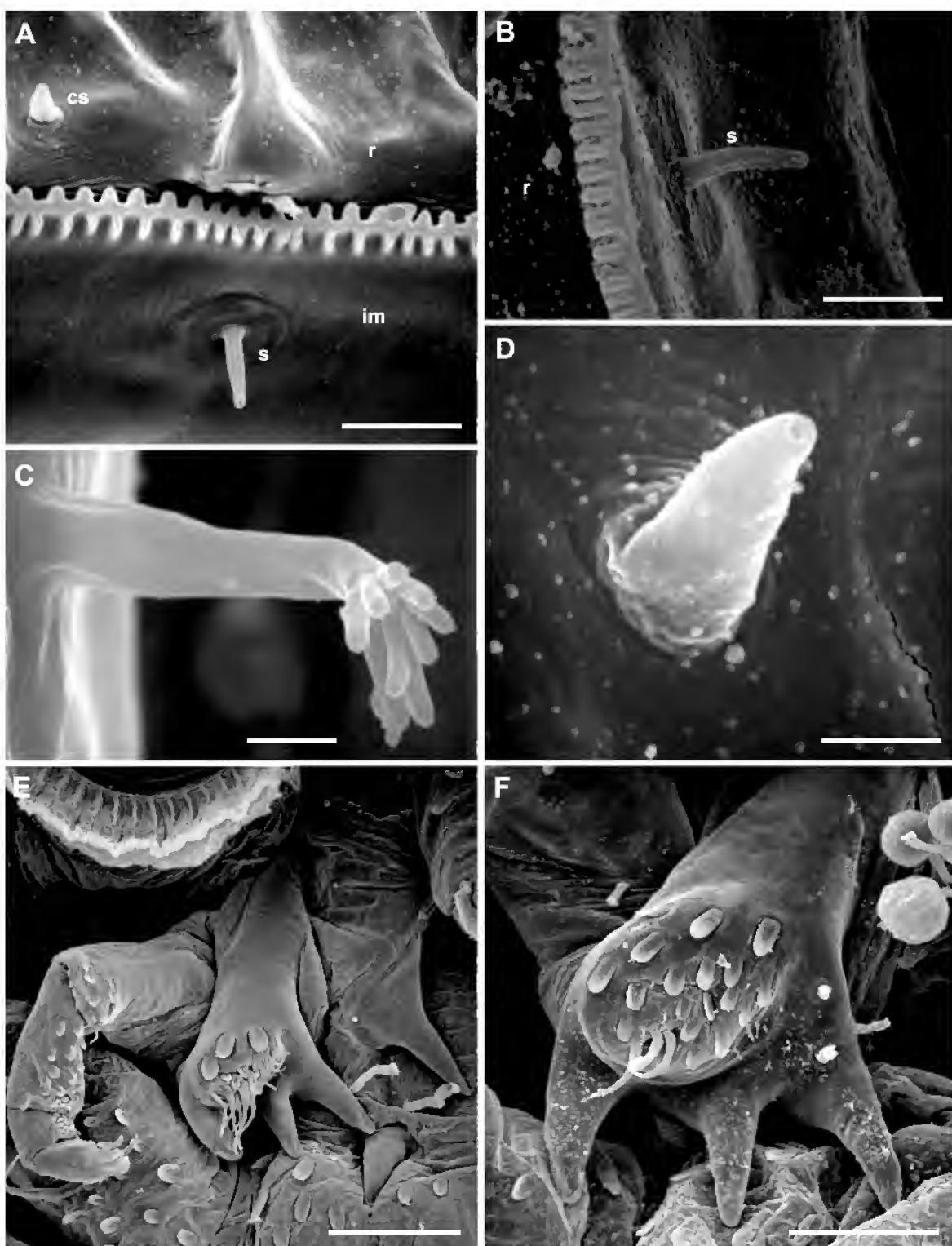
Figure. 6. *Argulus yucatanus*, n. sp. A) Conical sensillum (cs) and sensillum (s) on rim (r) and inner margin (im), respectively, of suction cup (male); note pore at tip of sensillum on inner margin of suction cup. B) Sensillum (s) on inner margin of suction cup; r = rim of suction cup (female). C) Sensillum with tentacles distally on inner margin of suction cup (male). D) Close-up of conical sensillum near basal sclerite on rim of suction cup (male); note pore at tip. E) Second maxilla, accessory spine, and post-maxillary spine (male, right side). F) Basal plate of 2nd maxilla and adjacent stalked protozoan parasites at upper right (female, right side). Scale: A, B = 5 μm; C, D = 1 μm; E, F = 100 μm.
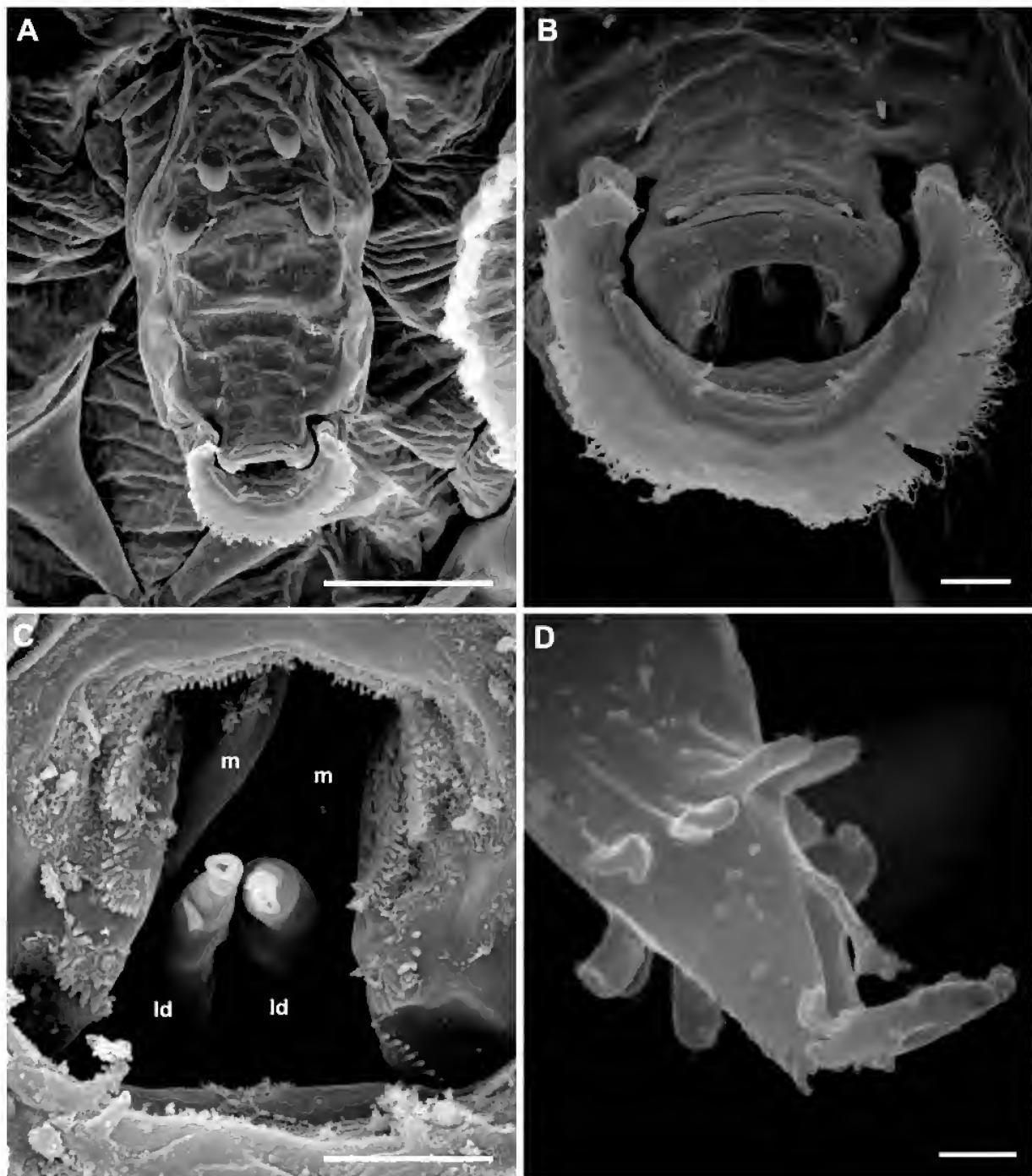
Figure. 7. *Argulus yucatanus*, n. sp. A) Mouth tube with 4 scales on basal portion (male). B) Mouth tube; note dense fringe of fine setae along margin of labium and sensilla on mouth tube, labium, and labrum (male). C) Labial ducts (ld) and mandibles (m) inside mouth (female); note openings at tips of labial ducts. D) Tip of preoral stylet with protuberances and opening (male). Scale: A = 50 μm; B, C = 10 μm; D = 1 μm.
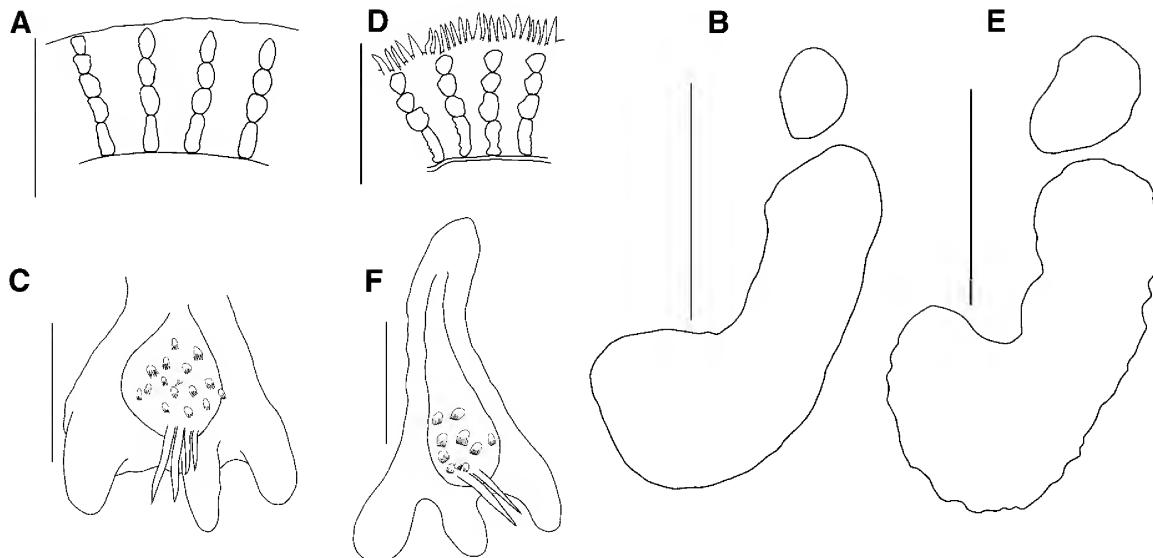
Figure 8. *Argulus chromidis* (holotype, female, ZMUC CRU-6030). A) Support rods from anterolateral portion of suction cup (fringe of setae not shown). B) Respiratory areas. C) Basal plate of left 2nd maxilla. *Argulus cubensis* (syntype, male, MCZ 8973). D) Support rods and fringe of setae from anterolateral portion of suction cup. E) Respiratory areas; note that scalloped appearance is an artifact of preservation and distortion. F) Basal plate of right 2nd maxilla. Scale: A, C, D, F = 100 μm; B, E = 500 μm.

characters of 2nd maxillae, shape of sclerites in suction cup support rods, or size and shape of spermathecae according to information given in the original description (see Wilson, 1936a); however, the type specimen of *A. rhamdiae* could not be located for direct comparison.

## DISCUSSION

Variation in number of and shape of sclerites was observed on *A. yucatanus* as was noted for several other species (Fryer 1959, Avenant-Oldewage and Oldewage 1995, Poly 2003; Figure 5). In *A. yucatanus* higher numbers of sclerites per rod occur in the anterolateral section of a suction cup and the sclerites tend to be more bulbous or round anterolaterally, whereas sclerite numbers are lower posteriorly and on mesial (inner) margin, and sclerites tend to be more slender. There also is a bilateral division of the suction cups that can be seen in the orientation of the rods, particularly the basal sclerites, and this has not been pointed out previously for any argulid. The orientation changes at 2 points, anterolaterally and posteromesially, and at one of these points, the thickened, uneven edges of the basal sclerites face one another, whereas at the opposite point, the thin edges face (Figure 5). This same type of change in orientation of the rods has been observed in other *Argulus* spp. (e.g., *A. cubensis*, Figure 8D), but not all species have this feature (W. Poly pers. obs.). For *A.*

*yucatanus* the number of rods on each "half" of a suction cup, as divided by the change in orientation, usually are not equal, with a slightly higher number occurring on the mesial side (means of 24 vs. 20). The bilateral division probably is an expression of normal bilateral symmetry of the body present in many metazoan phyla, although being slightly asymmetrical in this case. Cunnington (1931:262) illustrated 4 suction cup support rods of *Argulus carteri* Cunnington, 1931, showing the change in orientation at one point on a suction cup (his Plate 15, Figure 14), stating only that "... the apparently meaningless variations of these chitin rays afford additional evidence which it would be unwise to ignore."

*Argulus yucatanus* is the eighth *Argulus* species known to occur in Mexico. None of the other *Argulus* spp. that parasitize cichlids in the Gulf of Mexico and Caribbean region are similar to *A. yucatanus*. *Argulus chromidis* was described from a single female specimen, collected at Lake Nicaragua on the gills of a species of "*Chromis*," which Gill (1903) pointed out was likely a species of cichlid, whereas *Argulus cubensis* was discovered in Cuba on the cichlid, *Cichlasoma tetracanthus* (Krøyer 1863, Wilson 1936b). Structures on the bases of the 4th legs of *A. yucatanus* and *A. kosus* Avenant-Oldewage, 1994 appear to be nearly identical, and males of both species also have a fleshy lobe between the coxae of the 4th legs. However, there are numerous differences, e.g.,

body shape, features of the first and 2nd maxillae, and secondary sexual structures on legs of males, to name a few, that distinguish these 2 species (Avenant-Oldewage 1994, Van As et al. 1999). The pair of pegs on the 3rd legs of male *A. yucatanus* resemble structures on males of other species, including *A. arcassonensis* Cuénot, 1912, *A. kusafugu* Yamaguti and Yamasu, 1959, and *A. kosus* (Yamaguti and Yamasu 1959, Masson and Delamare Deboutteville 1962, Van As et al. 1999).

The parasites of *Cichlasoma urophthalmus* have been studied by Salgado-Maldonado and Kennedy (1997), Moravec et al. (1998), and Vidal-Martinez et al. (1998). *Argulus yucatanus* serves as an intermediate host of the nematode, *Mexiconema cichlasomae*, whose definitive host is *C. urophthalmus* (Moravec et al. 1999) and was abundant on *C. urophthalmus* in Celestun Lagoon in 1994 (Moravec et al. 1998). Argulids were reported as a component of the diet of *C. urophthalmus* in Celestun Lagoon (Martinez-Palacios and Ross 1988). Celestun Lagoon is estuarine/marine with variable salinities throughout (Martinez-Palacios and Ross 1992, Herrera-Silveira 1994). The lagoon contains a mixture of estuarine and marine fish species, and *C. urophthalmus* inhabits freshwater as well as brackish to marine habitats (Salgado-Maldonado and Kennedy 1997). Vidal-Martinez et al. (1998) reported *Argulus mexicanus* from *C. urophthalmus* in freshwater; possibly their specimens were not *A. mexicanus*, but rather, *A. yucatanus*.

There are noteworthy points to make concerning *A. funduli*. Wilson (1902) and others have indicated incorrectly that the description of *A. funduli* was based on a female specimen, but that the specimen illustrated was a male. In the original description, the figure legends were given in both Danish (p. 97 [p. 23 of separate]) and Latin (p. 412 [p. 338 of separate]) with the former referring to a male and the latter to a female. This author agrees that Krøyer's figure depicts a male specimen (Krøyer 1863, his Plate 2, Figure 1a). In addition, the original description includes remarks about the length of the carapace of both the male and the female, and the single lot of specimens registered as types of *A. funduli* contains 6 specimens (collected in the vicinity of New Orleans, Louisiana, USA), including both sexes, in the same state of preservation and representing one species (ZMUC CRU-6473). Therefore, after reading a complete translation of the original description, no doubt exists that the description was based on more than one specimen (both male and female), and all 6 specimens are syntypes. After examining the types of *A. funduli* and comparing them with other descriptions, published illustrations, and other specimens, it became quite clear that the name *A. funduli* has been applied incorrectly to other

species. Some illustrations of *A. funduli* in Wilson (1902), Meehean (1940), Cressey (1972), Kabata (1988), and Overstreet et al. (1992) appear to represent species other than *A. funduli*. Results of an investigation into the taxonomy of *A. funduli* will be reported elsewhere.

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# Fishery and Biology of Blackfin Tuna *Thunnus atlanticus* off Northeastern Brazil

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# FISHERY AND BIOLOGY OF BLACKFIN TUNA *THUNNUS ATLANTICUS* OFF NORTHEASTERN BRAZIL

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*ABSTRACT* Blackfin tuna, *Thunnus atlanticus*, is the target species of a handline artisanal fishery off northeastern Brazil in September–January, but it is also caught by anglers and as by-catch in industrial fisheries. The population structure, morphometric relationships, mortality, reproduction, and fishery dynamics were studied during 2 fishing seasons (1996 and 1997). The maximum length and weight observed were 87 cm FL and 10 kg $W_d$, respectively. Males were larger and predominant (1.9:1). The length at 50% maturity was 49.8 cm FL for females and 52.1 cm FL for males. This species uses the area for reproduction, although a spawning peak was not observed. The length at first capture (58.1 cm FL) was higher than the length at 50% maturity. The total, natural, and fishing mortality rates were 2.34, 0.94, and 1.40 $year^{-1}$, respectively. The total length-fork length and the total length-standard length relationships were TL = 1.35369 + 1.0462 FL and TL = 6.37742 + 1.0544 SL, respectively (sexes grouped). The length-weight relationship estimated for both sexes was $W_d = 0.00003\ FL^{2.8569}$. Annual catches decreased from 154 t $year^{-1}$ in the 1970s to 33.5 t $year^{-1}$ in the 1990s. It seems that there was not much change in the structure of this stock after 30 years, but the lack of a proper collection system of catch data and the increasing interest in recreational fisheries raise reasons for concern.

*RESUMEN* El Atún aleta negra, *Thunnus atlanticus*, es capturado por una pesquería artesanal en el noreste de Brasil de septiembre a enero, pero también por pescadores recreacionales y como fauna acompañante en pesquerías industriales. La estructura de la población, relaciones morfométricas, mortalidad, reproducción y dinámica pesquera fueron estudiadas durante dos temporadas de pesca (1996 y 1997). La longitud y el peso máximos observados fueron 87 cm FL y 10 kg $W_d$, respectivamente. Los machos fueron mayores y predominantes (1.9:1). El tamaño al 50% de madurez fue 49.8 y 52.1 cm FL para machos y hembras, respectivamente. Esta especie usa la región para reproducción, aunque un pico de desova no fue observado. El tamaño en la primera captura (58.1 cm FL) fue más alto que el tamaño al 50% de madurez. La mortalidad total, natural, y por pesca fueron 2.34, 0.94 y 1.40 $año^{-1}$, respectivamente. Las relaciones longitud total-longitud furcal y longitud total-longitud estándar fueron: TL = 1.35369 + 1.0462 FL y TL = 6.37742 + 1.0544 SL, respectivamente (sexos agrupados). La relación peso-longitud estimada para ambos los sexos fue de $W_d = 0.00003\ FL^{2.8569}$. Las capturas anuales disminuyeron de 154 t $año^{-1}$ en la década de los 70s a 33.5 t $año^{-1}$ en la década de los 90s. Los resultados parecen indicar que no ha habido mucho cambio en la estructura de este estoque después de treinta años, pero la carencia de un sistema apropiado de la obtención de datos de captura y el interés de las industrias pesqueras recreacionales son motivo de preocupación.

## INTRODUCTION

Blackfin tuna, *Thunnus atlanticus*, occur only in the western Atlantic Ocean, from Martha's Vineyard/US—40°N to Rio de Janeiro/Brazil—22°S, including the Gulf of Mexico and the Caribbean (Collette and Nauen 1983). Zavala-Camin (1991), however, recorded this species as far as 31°S (southern Brazil). Blackfin tuna reach a maximum size of 108 cm, which is much smaller than other *Thunnus* species (www.fishbase.org).

In the Caribbean region, this species supports important fisheries mainly in Venezuela, Martinique, Guadeloupe, Cuba, and Dominican Republic (FISHSTAT; www.fao.org). Off southern and northeastern Brazil, blackfin tuna are by-catch in longline fisheries targeting *Thunnus albacares*, *Thunnus obesus*, *Thunnus alalunga*, *Xiphias gladius*, and Carcarhinidae. Baía Formosa, a fishing village located in the south of Rio Grande do Norte State, is the only area in Brazil where significant aggregations of *T. atlanticus* are found close enough to the mainland to justify a handline artisanal fishery from September to January (Cruz and Paiva 1964). This fishery is important to the local economy, where almost 100% of the artisanal catch of 'albacore' is *Thunnus atlanticus*. Catches are consumed locally or sold to neighbour states: Paraíba and Pernambuco (Tartari 1966). The artisanal fishery targeting blackfin tuna in Brazil cannot be properly analyzed because national and local databases record this species as albacore ('albacora') together with 3 other species (*T. albacares*, *T. alalunga*, and *T. obesus*). Indeed, Freire

(2003), in compiling a national landing database, noted that albacore is caught by both artisanal and industrial fisheries in 14 out of the 17 Brazilian coastal states. However, blackfin tuna ('albacorinha') is recorded as being caught only after 1994 by industrial fisheries in 3 states in southern Brazil (São Paulo, Rio de Janeiro, and Santa Catarina). No catch is recorded for this species in northeastern Brazil.

Information on the biology and fishery of blackfin tuna is available primarily for the Caribbean (Carles 1974, Garcia-Coll 1987a, Carles and Valle 1989, Báez-Hidalgo and Bécquer 1994, Taquet et al. 2000, Doray et al. 2004) and for Brazil in earlier periods (Cruz and Paiva 1964, Monte 1964a, b, Cruz 1965, Nomura and Cruz 1966).

This study aims to update information on the fishery and population structure of *Thunnus atlanticus* off northeastern Brazil. In particular, we document overall catches, yield per boat per trip, length-frequency distribution, sex ratio, basic morphometric relationships, mortality rates, and length at 50% maturity and at first capture.

## MATERIALS AND METHODS

A sampling program was established in Baía Formosa, Rio Grande do Norte/Brazil (6°22'S and 35°00'W; Figure 1), during September 1996–January 1997 and September 1997–January 1998, the first sampling period following more than thirty years without data collection from commercial fisheries—artisanal or industrial. This program was divided into 2 parts: size sampling, where measures of fork length (cm; FL) and gutted weight (kg; $W^d$ = weight with no viscera or gills) of blackfin tuna *Thunnus atlanticus* were taken daily, and biological sampling, where we measured weekly total length (cm; TL), FL (cm), and $W^d$ (kg), determined sex, and collected gonads.

All samples were taken in the 5 market places concentrated in the only landing port of the region. In both sampling programs, all individuals caught by each boat were sampled up to an overall total of 30 fishes per sampling day (some boats catch daily only 2–3 individuals). This dataset was complemented with TL and standard length (SL) data from the REVIZEE Program/NE Score (Assessment of Renewable Resources off the Brazilian Exclusive Economic Zone/Northeastern Score) for the period 1998–2000. Catch data per boat per trip for blackfin tuna were obtained from receipts available at the Baía Formosa Fishing Cooperative (1996–1998) and complemented with data from national and local statistical bulletins (CEPENE, 2000; Freire, 2003).

Frequency distributions of FL were calculated for males and females. TL-FL and $W^d$-FL relationships were estimated for males and females separately, with a log-transformation of $W^d$ and FL for the latter. A SL-TL relationship was calculated for males and females combined, as sex information was not available for specimens with recorded SL.

The total instantaneous mortality rate was estimated based on the catch-curve for FL converted to age using von Bertalanffy growth parameters for blackfin tuna estimated using the routine ELEFAN I—Electronic Length-Frequency Analysis available in FISAT II—FAO-ICLARM Stock Assessment Tools (http://www.fao.org/fi/statist/fisoft/fisat/index.htm; Gayanilo and Pauly, 1997). This routine identified the growth curve that best fitted the set of length-frequency data for blackfin tuna obtained from the size sampling previously described. The natural mortality rate (M) was estimated based on the following simplified equation:

$$M = K^{0.6543} \cdot L_\infty^{-0.279} \cdot T_c^{0.463}$$

where: $L_\infty$ = asymptotic length (TL; cm), K = curvature parameter of the von Bertalanffy growth curve (year$^{-1}$), and Tc = mean water temperature (°C) (Pauly 1980). A mean sea-surface temperature of 27.2 °C was estimated for the fishing seasons of 1996 and 1997, based on the data available from the International Comprehensive Ocean-Atmosphere Data Set—ICOADS (http://dss.ucar.edu/pub/coads/). The mean size at first capture was estimated by fitting a logistic curve to the ascending limb of the length-converted catch curve and defining the size at which 50% of the individuals are caught ($L_{50}$).

Maturity stages were defined macroscopically using the following scale: 1 = immature; 2 = resting; 3 = active; 4 = ripe (actual spawning condition); and 5 = spent (Jolley 1977). The weight of the gonads was measured (0.001 g) and the gonadosomatic index (GSI) was calculated for females and males as: GSI = ($W_g$ x 100)/$W_d$, where $W_g$ = gonadal weight (g) and $W_d$ = gutted weight (g). The length at maturity was estimated for females and males as the length at which 50% of all individuals are mature. It is worth mentioning here that, although individuals are eviscerated on board (Nomura and Cruz 1966), this process involves removal of gills and viscera through the operculum, leaving the gonads intact. Hence, it is still possible to determine the sex and reproductive condition of eviscerated fish.

Length-frequency distributions for males and females were compared using a Chi-square test. Sex ratios were tested against the null hypothesis of 1:1 for each month using a Chi-square test. Length-weight and length-length relationships for males and females were compared using a t-statistic to test for both slope and intercept (Zar, 1984).

**Figure 1. Location of sampling areas in northeastern Brazil. The black circle corresponds to Baía Formosa (6°22'S and 35°00'W), and the gray circles are areas where samples were collected through the REVIZEE Program/Score NE (MA = Maranhão, PI = Piauí, CE = Ceará, RN = Rio Grande do Norte, PB = Paraíba, PE = Pernambuco, AL = Alagoas, SE = Sergipe, BA = Bahia). RJ = Rio de Janeiro, SP = São Paulo, SC = Santa Catarina. In the bottom-right are the countries with the highest catches of blackfin tuna in the Caribbean.**

All statistical tests were performed with a significance level of 0.05.

## RESULTS

From 1993 to 2001, blackfin tuna catches from the artisanal fleet operating in Baía Formosa ranged from 16.8 to 48.6 tonnes, with an annual mean of 33.5 tonnes. This estimate was obtained considering that 100% of 'albacore' catches in that region are actually blackfin tuna. The mean catch of blackfin tuna per boat per fishing trip was 35.3 kg

for 1996–1997, with an increasing trend towards the end of the fishing season when the yield reached 50.7 kg. Length-frequency distributions for males and females were statistically different ($x^2 = 71.6$; DF = 23; $P < 0.0001$), with males reaching a larger size (Figure 2).

Analysis of monthly length-frequency distributions indicates that larger individuals reach the region in September and are followed later by a second mode of smaller individuals (Figure 3). The smallest observed individual was 23 cm FL and the smallest observed weight was 0.8 kg $W_d$ (Table 1). The maximum length and weight
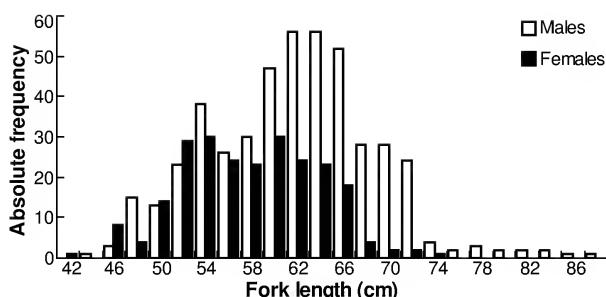
**Figure 2. Length-frequency distribution for *Thunnus atlanticus* off Baía Formosa, northeastern Brazil (1996–1998). $n_{males}$ = 458; $n_{females}$ = 237.**

observed were 87 cm FL, and 10 kg $W_d$, respectively. Males predominated in a ratio of 1.9:1 (Table 2). Although this proportion varied during the fishing season, males were always predominant as indicated by the chi-square test, with the exception of the sample collected in January 1997. In this sample, the sex ratio was not significantly different from 1:1.

The relationships between TL and FL for females and males were not statistically different ($t_{slope}$ = 1.70, DF = 336, and $P$ = 0.090; $t_{intercept}$ = 1.68, DF = 337, and $P$ = 0.094). The resulting relationship obtained for both sexes combined was: TL = 1.35369 + 1.0462 FL (Table 3). The relationship between TL and SL estimated for the sexes combined was: TL = 6.37742 + 1.0544 SL (Table 3). The relationships between $W_d$ and FL for males and females were statistically different, with males heavier at size (Table 3; $t_{slope}$ = 2.16, DF = 613, and $P$ = 0.031; $t_{intercept}$ = 2.24, DF = 614, and $P$ = 0.025). The weight-length relationship for unsexed individuals from our sampling area is: $W_d$ = 0.00003 $FL^{2.8569}$ ($r^2$ = 0.92, $n$ = 617, $P$ < 0.0001).

The von Bertalanffy growth curve parameters estimated based on the length-frequency distribution was: $L_{\infty}$ = 92 cm (FL), K = 0.65 $year^{-1}$, $t_0$ = 0 years. The total instantaneous mortality rate (Z) estimated was 2.34 $year^{-1}$ (Confidence interval (CI) = [1.92;2.77]; Figure 4). The natural mortality rate calculated was 0.94 $year^{-1}$, which implies a fishing mortality of 1.40 $year^{-1}$ and an exploitation rate of 59.7%. The average length at first capture was 58.1 cm FL (sexes grouped; Figure 5).

The length at 50% maturity was greater for males (52.1 cm FL) than for females (49.8 cm FL) (Figure 6). The gonadosomatic index (GSI) for females did not show a clear pattern between years (Figure 7). For males, highest values of GSI were observed in December in both years. Macroscopic identification of maturity stages for females indicates that this species uses the area for reproduction, with active individuals observed as early as

September or October (Figure 8). One month later running ripe individuals were observed, with some inter-annual variation.

## DISCUSSION

Blackfin tuna catches from Baía Formosa were much lower in 2001 (48.6 tonnes) than in 1969–1977 (52 to 296 tonnes, with an average of 154 tonnes $year^{-1}$; Vasconcelos and Conolly 1980). Current catches are also lower than catches from industrial fisheries off southeastern Brazil (annual average of 172 tonnes $year^{-1}$ for 1995–2000; Freire 2003). However, the social importance of the artisanal fishery is higher, as the local community largely depends on this fishery.

The mean yield per boat per fishing trip in 1963 was 34.9 kg (Cruz and Paiva 1964b). The yield increased to 39.3 kg in 1977 (68 boats, Vasconcelos and Conolly 1980) but decreased to 38.5 kg in 1996 (this study). These differences cannot be attributed to changes in gear selectivity; 80–140 cm long handline has been used for the last 40 years with a no. 15 hook attached to the end of the line (Vasconcelos and Conolly 1980; João C. Neto, Baía Formosa Fisher's Association, pers. com.). The only change observed during this period was the introduction of motorized boats in 1967–1968, which could account for the yield per trip being higher in the 1970s than in the early 1960s. Although these boats operate in the same fishing ground as sailboats, 12 to 16 miles from the coast (Vasconcelos and Conolly 1980), their fishing trips are longer (3–5 days versus 1 day for sailboats) and their crew is larger (4 fishers versus 3) (João C. Neto, pers. com.), thus producing higher yield per trip.

Total blackfin tuna mortality decreased from 3.16 $year^{-1}$ in 1965 (based on Nomura and Cruz 1966) to 2.66 $year^{-1}$ in 1977 (based on Vasconcelos and Conolly 1980) and declined again to 2.34 $year^{-1}$ in 1996, even though the latter two are not statistically different. More effort should be put into the collection of catch and effort data for this fishery since fishing mortality is estimated at 1.40 $year^{-1}$, with an exploitation rate of about 60%, which may not be sustainable. On the other hand, a length at first capture greater than the length at 50% maturity may contribute to the future sustainability of this fishery. Although the natural mortality obtained in this study is similar to those for other *Thunnus* species, there could be a size-dependence as pointed out by Hampton (2000) for *T. albacares* and *T. obesus*.

Blackfin tuna use the Baía Formosa area for reproduction, although we could not define a distinct reproductive peak using GSI or macroscopically-defined maturity
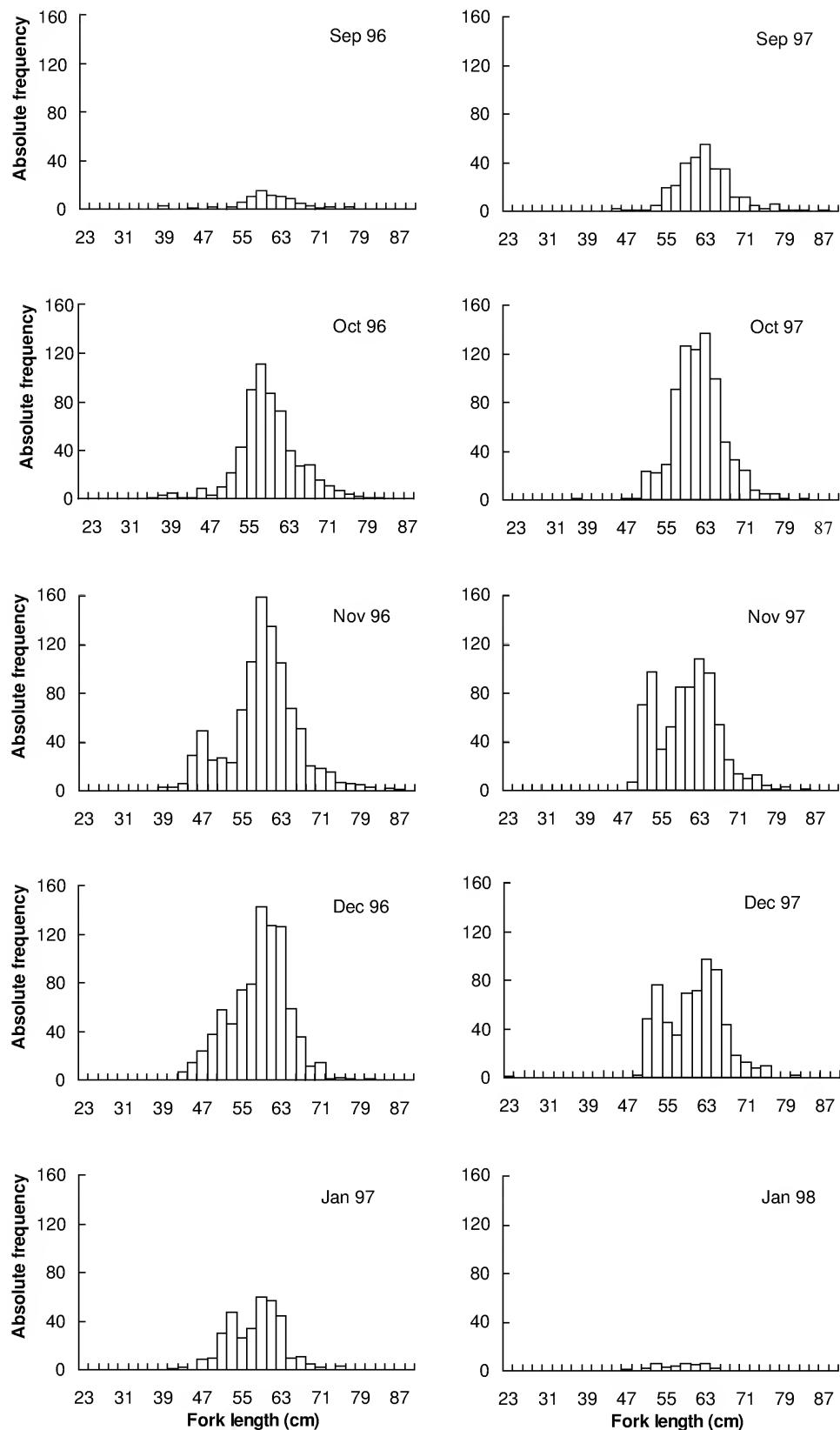
**Figure 3. Monthly length-frequency distribution for *Thunnus atlanticus* off Baia Formosa (1996–1998; both sexes combined; *n* = 5315).**

## TABLE 1

Minimum, mean, and maximum size of *Thunnus atlanticus* off Baía Formosa sampled from September 1996-January 1998. TL = total length (cm), FL = fork length (cm), $W_d$ = gutted weight (kg), SD = standard deviation, N = sample size.

| | Unsexed | | | Females | | | Males | | |
|---|---|---|---|---|---|---|---|---|---|
| | TL | FL | $W_d$ | TL | FL | $W_d$ | TL | FL | $W_d$ |
| Minimum | 50.0 | 23.0 | 0.8 | 50.0 | 39.5 | 1.5 | 53.0 | 43.0 | 1.5 |
| Mean | 63.8 | 59.4 | 3.3 | 61.2 | 56.1 | 2.7 | 65.2 | 60.2 | 3.4 |
| Maximum | 90.5 | 87.0 | 10.0 | 74.5 | 72.5 | 5.5 | 90.5 | 86.0 | 9.5 |
| SD | 6.9 | 6.2 | 1.2 | 5.9 | 5.9 | 0.8 | 7.0 | 7.3 | 1.2 |
| N | 357 | 5316 | 5209 | 110 | 237 | 110 | 230 | 457 | 230 |

## TABLE 2

Number of males and females and sex ratio of *Thunnus atlanticus* caught off Baía Formosa by month. Jan 1998 was not included as there was no biological sampling. *Statistically significant at = 0.05.

| Month | Males | Females | Sex ratio | Chi-square ($^2$) | p |
|---|---|---|---|---|---|
| Oct 1996 | 30 | 15 | 2.0:1 | 5.0* | 0.0253 |
| Nov 1996 | 94 | 53 | 1.8:1 | 11.4* | 0.0007 |
| Dec 1996 | 67 | 31 | 2.2:1 | 13.2* | 0.0003 |
| Jan 1997 | 26 | 20 | 1.3:1 | 0.8 | 0.3771 |
| Sep 1997 | 33 | 16 | 2.1:1 | 5.9* | 0.0151 |
| Oct 1997 | 104 | 47 | 2.2:1 | 21.5* | < 0.0001 |
| Nov 1997 | 80 | 44 | 1.8:1 | 10.4* | 0.0012 |
| Dec 1997 | 24 | 11 | 2.2:1 | 4.8* | 0.0280 |
| Total | 458 | 237 | 1.9:1 | 70.3* | < 0.0001 |

## TABLE 3

Length-length and weight-length relationships for blackfin tuna *Thunnus atlanticus* off northeastern Brazil. TL = total length (cm); FL = fork length (cm); SL = standard length (cm); $W_d$ = gutted weight (kg); N = sample size; $r^2$ = coefficient of determination; p = probability. * Linear relationship: Y = a + bX; ** Power relationship: Y = aX$^b$.

| Unknown | Known | a | b | Sex | N | $r^2$ | p |
|---|---|---|---|---|---|---|---|
| TL* | FL | 0.21206 | 1.0678 | Females | 110 | 0.98 | < 0.0001 |
| TL* | FL | 1.48854 | 1.0433 | Males | 230 | 0.99 | < 0.0001 |
| TL* | FL | 1.35369 | 1.0462 | Unsexed | 340 | 0.99 | < 0.0001 |
| TL* | SL | 6.37742 | 1.0544 | Unsexed | 93 | 0.90 | < 0.0001 |
| Wd** | FL | 0.00004 | 2.7268 | Females | 218 | 0.86 | < 0.0001 |
| Wd** | FL | 0.00002 | 2.8837 | Males | 399 | 0.94 | < 0.0001 |
| Wd** | FL | 0.00003 | 2.8569 | Unsexed | 617 | 0.92 | < 0.0001 |
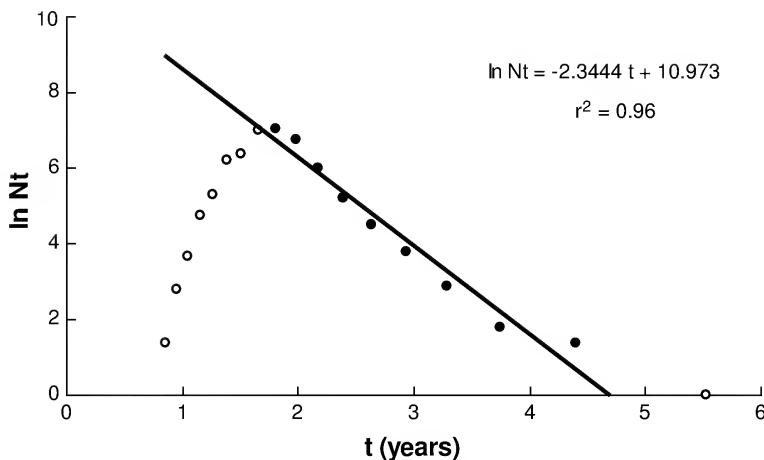
**Figure 4. Length-converted catch-curve for *Thunnus atlanticus* off Baía Formosa (1996–1998; both sexes combined). Confidence interval (CI) for total instantaneous mortality (Z) = [1.92; 2.77].**

$$\ln Nt = -2.3444\, t + 10.973$$
$$r^2 = 0.96$$
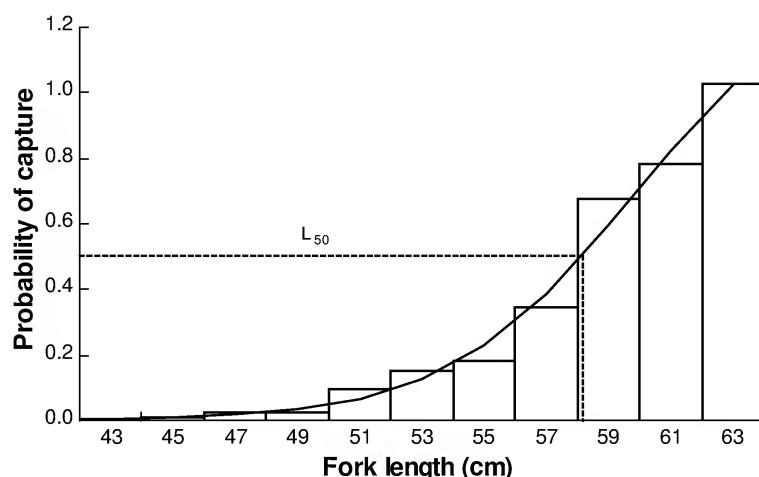


**Figure 5. Probability of capture by handline for *Thunnus atlanticus* off Baía Formosa (1996–1997; both sexes combined; $L_{50}$ = 58.1 cm FL).**
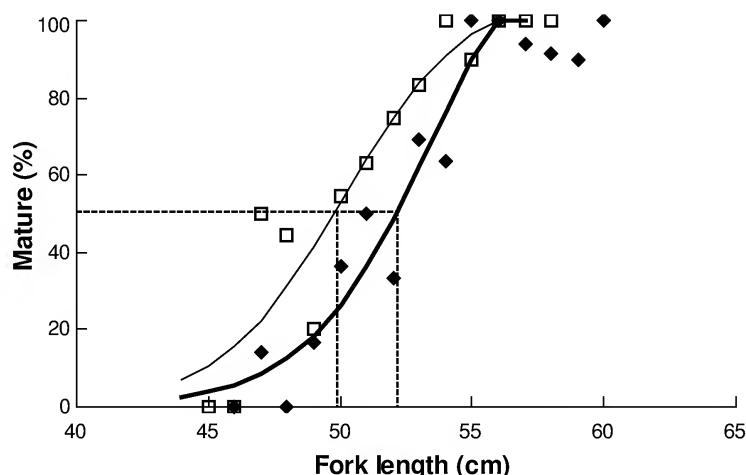


**Figure 6. Sexual maturity for females (open squares) and males (solid diamonds) of *Thunnus atlanticus* off Baía Formosa (1996–1998; $n_{females}$ = 223; $n_{males}$ = 432). Dashed lines represent length at 50% maturity (49.8 and 52.1 cm FL for females and males, respectively).**
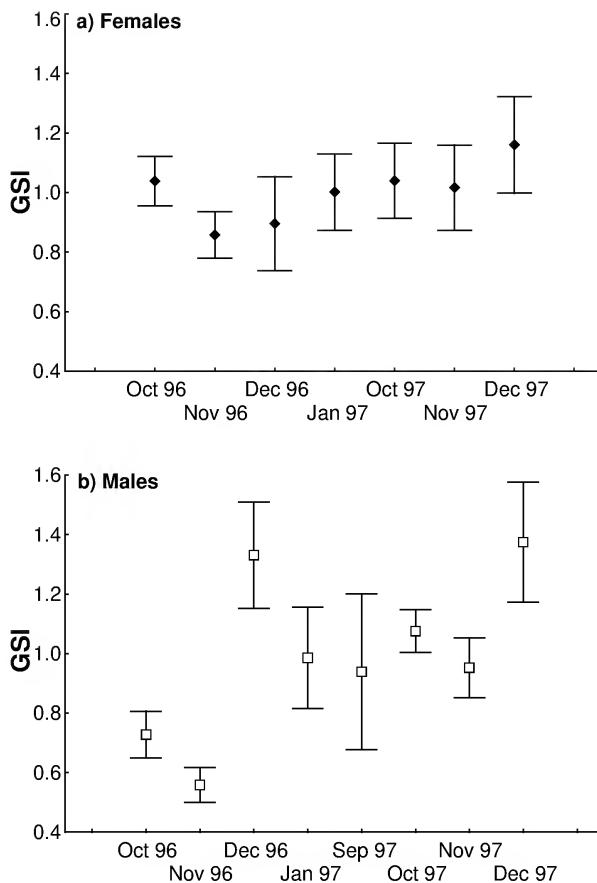
Figure 7. Variation of the mean gonadosomatic index (GSI) for females (a) and males (b) of *Thunnus atlanticus* off Baía Formosa (1996–1997; $n_{females}$ = 177; $n_{males}$ = 293). Whiskers represent mean ± standard error.

stages. In December, GSI for males was higher than for females, which is not common for tunas, but was also found in *T. obesus* in the Indian Ocean (Nootmorn 2004). The lack of such a peak of reproduction may be due to the pattern of immigration of this species to this region, with large individuals arriving first, followed by smaller individuals, and/or to its multiple spawning feature, commonly observed in tunas (Schaefer 2001). This author points out that sea-surface temperatures (SST) higher than 24 °C are associated with spawning activity for all tuna species. Indeed, local SST was in excess of 26 °C during the whole sampling period (http://dss.ucar.edu/pub/coads/), which would reinforce the hypothesis of continuous spawning activity. However, there is no information on the occurrence of larvae of *T. atlanticus* in this region, probably due to identification difficulties for *Thunnus* larvae (Richards et al. 1990). In contrast to Brazil, spawning occurs year round in Cuba, with a clear peak in June–September

(Valle-Gomez 1992). In southeastern US, spawning occurs from April to November (Idyll and de Sylva 1963).

Males are larger than females, as also observed for nearly all tuna species (Schaefer, 2001), and mature at a slightly larger size. Although a larger maturity size for males is not common in scombrids, length at 50% maturity for *Katsuwonus pelamis* in both Atlantic and Indian oceans was reported to be larger for males than for females (Cayré and Farrugio 1986, Stéquert and Ramcharrun 1996). A detailed histological study would be able to test if this difference is real or possibly attributed to the misclassification of maturity stages using macroscopic analysis.

Monte (1964b) pointed out that sexual maturation for both sexes of blackfin tuna occurring in northeastern Brazil in the early 1960s begins at 50 cm FL, with a higher frequency of ripe females at 56–65 cm FL. These values are higher than the size at 50% maturity estimated in this study. Such reduction in maturity size could result in shorter reproductive life span and reduced fecundity, even though a compensatory increase in mean individual fecundity has been observed for some species (Jennings et al. 2001). There are no conclusive data on the length at 50% maturity for the Caribbean, except for an indication of sexual differentiation occurring in individuals 39 cm long in Cuba (Carles 1971) and an indication that individuals smaller than 41 cm are immature in Martinique (Taquet et al. 2000).

Males were predominant off Baía Formosa in 1996 (1.9:1) as they were in the 1960s (1.6:1; Monte 1964b), with some variation during the fishing season. A predominance of males was also observed off Cuba (1.6:1, Garcia-Coll 1987b) and off Miami (2:1, Idyll and de Sylva 1963). Although the overall sex ratio for most tuna species is 1:1, some concentration of males occur when females are reproductively active (Schaefer 2001), as observed here.

In addition to the apparent reduction in the length at 50% maturity for females, their mean size also decreased by about 1.5 cm FL in 34 years. However, smaller and bigger individuals were sampled in 1996–1997 (23–87 cm FL; this study) than in 1963–64 (51–80 cm FL, Monte 1964b), 1965–1966 (45–79 cm FL, Nomura and Cruz 1966) and 1977 (36.5–81.5 cm FL, Vasconcelos and Conolly 1980). These observed changes cannot be attributed to gear selectivity, as gear has remained the same during this period. Instead, they indicate actual changes in the population and represent signs of intense exploitation, as observed, for example, for *Micropogonias undulatus* (Atlantic croaker) in the northwestern Atlantic (Diamond et al. 1999). However, because *T. atlanticus* is a migratory species, these effects are likely to be a result of local
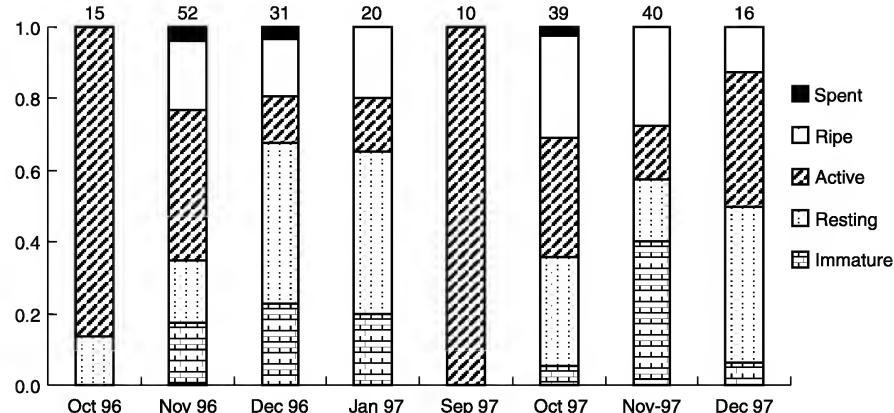
Figure 8. Proportion of the stages of maturity for females of *Thunnus atlanticus* off Baía Formosa (1996–1997). Sample sizes are presented on the top of each column.

exploitation combined with that from other areas in the distribution range of the stock.

Some changes have been noted in the population structure, in catches, and in yield of blackfin tuna. Although these changes are not large enough to cause great concern, they should be seen as a warning sign by the national agencies in charge of fisheries management (SEAP—Special Secretary of Aquaculture and Fisheries and IBAMA—Brazilian Institute for the Environment and Renewable Resources) and should promote improved data collection for artisanal fisheries targeting small tunas. The importance of blackfin tuna in the food web of large pelagic fish is not well understood, as it is often difficult to identify species or even genera of scombrids found in the guts of billfish, swordfish, dolphinfish, and sharks (Vaske-Júnior 2000). There is also an increasing demand for oceanic recreational fisheries in the region (Freire in press), which could ultimately put more pressure on this resource. Thus, it is imperative that our understanding of blackfin tuna be improved sooner rather than later.

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# Gulf and Caribbean Research

Coastal Origin of Common Snook, *Centropomus undecimalis,* in Florida Bay

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# COASTAL ORIGIN OF COMMON SNOOK, *CENTROPOMUS UNDECIMALIS*, IN FLORIDA BAY

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ABSTRACT We used the elemental signatures of otoliths to investigate the coastal origin of common snook (*Centropomus undecimalis*) in Florida Bay, Florida and evaluate current management boundaries. We examined juvenile otoliths from Florida's Atlantic and Gulf of Mexico (Gulf) populations and determined that there were significant differences in several elemental ratios (Mn/Ca, Cu/Ca, Sr/Ca, Ba/Ca). In addition, a discriminant function analysis (DFA) indicated a significant separation between the juveniles from each coast and otoliths were never misclassified by coast, indicating a distinct difference in their otolith chemistry. Using only juvenile otoliths to derive a calibration function, a separate DFA indicated that the adults from Florida Bay likely originated from both coasts of Florida in roughly equal proportions. Although these preliminary results contradict tagging studies, they concur with genetic studies suggesting that both east and west coast populations contribute to the common snook found in Florida Bay.

## INTRODUCTION

The effective management of marine species requires some knowledge of the source of recruits to the population. Despite the importance of such information, discerning the origin of individuals can often be quite difficult, as many marine species have larvae or juveniles that can widely disperse, thereby creating demographically open populations (Roughgarden et al. 1988). Conventional techniques such as genetics and mark-recapture have often proven inadequate in identifying recruitment source either due to low resolution (e.g., < 1% exchange renders populations genetically homogeneous; Kimura and Maruyama 1971) or logistical problems (e.g., tagging and recapturing larvae/juvenile that can disperse vast distances and suffer high mortality; Thorrold et al. 2002). In this paper we examine the issue of the coastal origins of common snook, *Centropomus undecimalis*, an economically and ecologically important species, using otolith chemistry.

Common snook are long-lived (21 years), late-maturing (4–5 years) protandric hermaphrodites that are distributed along the coasts of Florida's Atlantic Ocean (Atlantic) and Gulf of Mexico (Gulf) (Taylor et al. 1998, 2000). This gamefish supports valuable sport fisheries throughout its range and contributes substantially to Florida's economy (Tucker et al. 1985). Adult common snook support popular fisheries in the Florida Keys and adjoining Everglades National Park (Figure 1, Tilmant et al. 1989), but the source of recruits to this area remains unknown. Several studies have reported collecting common snook in Florida Bay (Tabb and Manning 1961, Tabb et al. 1962, Roessler 1970). However, none recorded the sizes of the individuals and it is likely that these records are of adults because of the high salinities of the waters in which they were collected. No eggs, larvae, or juvenile common snook were found in several other studies in Florida Bay (Rutherford et al. 1986, Collins and Finucaine 1987, Powell et al. 1989, Ley et al. 1999), suggesting that the major source of recruitment to the adult stock in this region originates elsewhere.

Tringali and Bert (1996) examined the genetic stock structure of common snook throughout its range and found that Atlantic and Gulf populations were reproductively isolated. Their data showed that adult common snook from the western portion of Florida Bay exhibit transitional properties of both populations and suggested that adult common snook in that area were recruited from both coasts of Florida. Tagging studies, however, have indicated that the Atlantic population is the most likely source of common snook in Florida Bay (Peters 1993, Bruger and Whittington, unpublished data).

Water masses vary in their chemical composition in both time and space. During otolith growth, elements from seawater can substitute for calcium in the otolith matrix (Campana 1999). Thus, otoliths have the potential to act as natural tags. Otolith trace element signatures have been useful in delineating stocks (Campana et al. 1994, Patterson et al. 1999, 2004), distinguishing juvenile nursery areas (Gillanders and Kingsford 2000, Forrester and Swearer 2002), and examining natal homing and self-recruitment (Swearer et al. 1999, Thorrold et al. 2001).

The objective of this study was to further investigate the coastal origin of common snook in Florida Bay as considerable research effort has yet to provide a clear understanding of the source of adult common snook in the Florida Bay assemblage. In addition, we wanted to evaluate the current fisheries management boundaries of com-
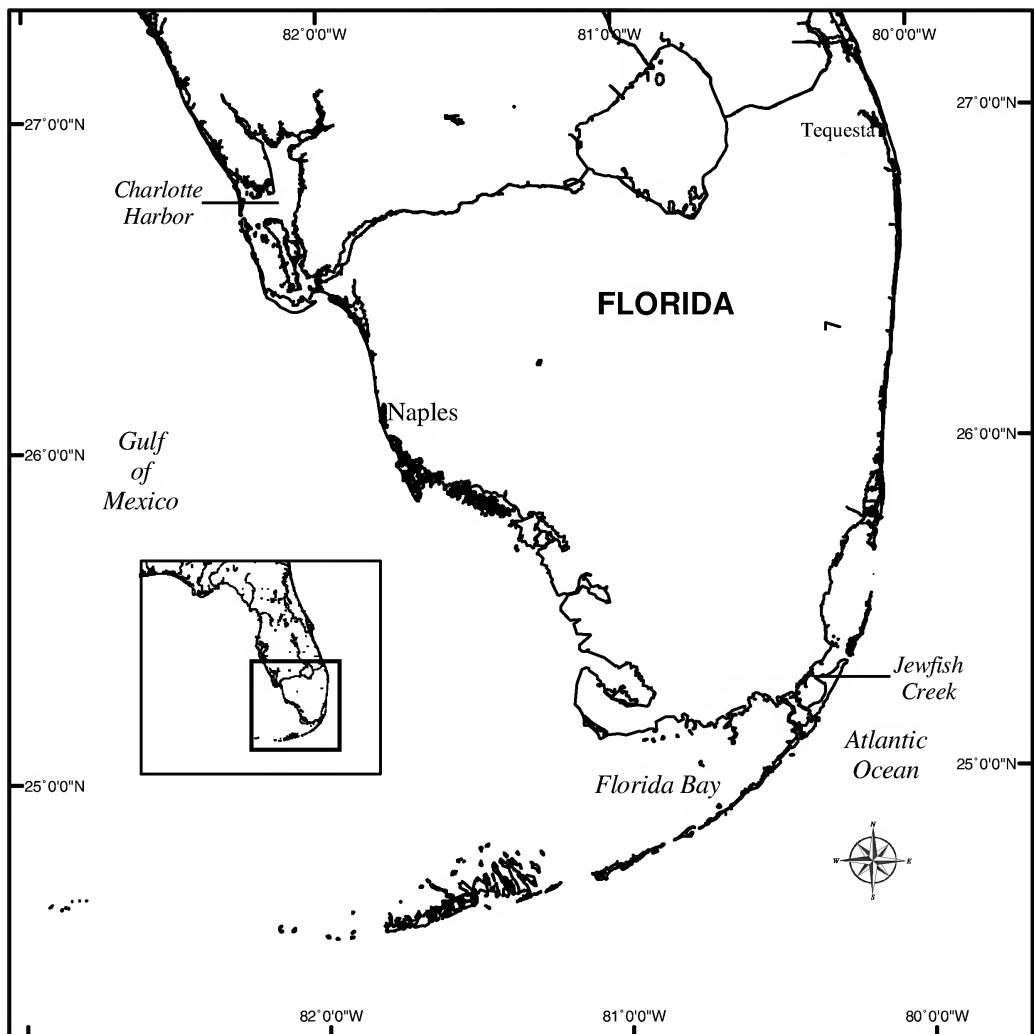
**Figure 1. Map of Florida depicting the three sampling locations: Charlotte Harbor (CH), Tequesta (TQ), and Florida Bay (FB).**

mon snook in Florida Bay based on our findings. We chose to take a new approach to the issue of common snook origin and examined the chemistry of juvenile common snook otoliths from both Atlantic and Gulf populations, as well as the otolith cores from adult common snook in Florida Bay. Because otolith chemistry primarily reflects the chemistry of the water in which the fish resides (Bath et al. 2000), the elemental composition of the cores of adult common snook otoliths should bear the signatures of their natal estuary. The results of this type of investigation may identify not only the coastal origin of the recruits to Florida Bay, but also may be used to quantify the relative contributions of Atlantic and Gulf stem populations if mixing of these populations occurs.

## METHODS

### Sample collection

Young-of-the-year common snook ($n = 20$ per location; 93–250 mm SL) were collected by seine and hook and line during March–July 1999 in the vicinity of Tequesta and Charlotte Harbor on the Atlantic and Gulf coasts of Florida; these 2 locations represented Atlantic coast and Gulf coast common snook populations (Figure 1). The common snook were frozen whole until all the samples from these 2 locations were collected. Adult common snook ($n = 20$; 306–615 mm SL) from northeastern Florida Bay were captured during September–December 1999 (Figure 1). The otoliths were removed in the field, rinsed, and stored dry. Due to the limited number of adults available, it was not possible to match adults and juveniles by year class.

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## TABLE 1

**Mean elemental ratios (± SE) in the otoliths of common snook, *Centropomus undecimalis*, from each of the 3 sampling locations (*n* = 20). Ratios are given in µmol/mol Ca.**

| Elemental Ratio | Florida Bay | Tequesta | Charlotte Harbor |
|---|---|---|---|
| Mg/Ca | 125.70 ± 4.73 | 122.74 ± 5.08 | 130.92 ± 5.12 |
| Mn/Ca | 4.54 ± 0.41 | 2.88 ± 0.20 | 4.94 ± 0.30 |
| Cu/Ca | 0.26 ± 0.022 | 0.16 ± 0.0078 | 0.19 ± 0.013 |
| Zn/Ca | 0.72 ± 0.099 | 0.56 ± 0.060 | 0.52 ± 0.034 |
| Sr/Ca | 2249 ± 115 | 3094 ± 30 | 3865 ± 66 |
| Ba/Ca | 2.18 ± 0.29 | 1.40 ± 0.41 | 2.65 ± 0.3 |

## Sample preparation and analysis

Sample preparation and analysis procedures are similar to those described in Patterson et al. (2004). Juvenile otoliths were polished evenly on all sides with 220-grit size lapping paper until the remaining core section weighed about 10 mg. Adult otoliths from Florida Bay were first sectioned with a Buehler Isomet low-speed saw and were then polished using the method described above. The weights of the otolith sections used in the analysis did not differ (ANOVA, $F_{2,57} = 2.37$, $P > 0.05$). To remove surface contamination, all the sections were then acid washed in 1% ultrapure HNO3 for 15 seconds and triple-rinsed in Milli-Q water. They were then dried in a class 100 laminar flow hood for 24 h and weighed to the nearest 10 µg. The otoliths were then placed in 0.5 ml of 70% ultrapure $HNO_3$ and dissolved for analysis. The final volume was brought up to 5 ml with Milli-Q water. Blanks were prepared in the same manner to calculate limits of detection (LOD) and for blank corrections.

Elemental concentrations of the otoliths were determined using a Perkin-Elmer Elan 5000 inductively coupled plasma mass spectrometer (ICP-MS). Preliminary tests indicated that 7 elements ($^{26}$Mg, $^{55}$Mn, $^{43}$Ca, $^{63}$Cu, $^{66}$Zn, $^{86}$Sr, and $^{138}$Ba) were detectable and suitable for ICP-MS analysis. Sample order was blocked so that one otolith from each location was sampled in turn, with the order within each block randomized. Internal standards for each element were used and referenced against $^{45}$Sc, $^{72}$Ge, $^{89}$Y, and $^{159}$Tb. Instrument drift was monitored by analyzing a calibration verification solution every 20 samples; acceptable recovery was ±10% of the expected value. Precision was typically < 5% relative standard deviation (RSD) for Ca and Sr and < 10% for trace elements. The LOD for each element was calculated from the prepared blanks as 3 plus the mean blank value with the following results (in µg g$^{-1}$): $^{43}$Ca 126, $^{86}$Sr 0.43, $^{138}$Ba 0.04, $^{26}$Mg 0.29, $^{55}$Mn 0.08, and $^{66}$Zn 0.04, $^{63}$Cu 0.03. Observed values were well above the LOD.

## Statistical analysis

Elemental data were standardized to Ca and expressed as molar concentrations. The assumption of homogeneity of variances in elemental data was tested using a Cochran's C-test and data were subsequently ln (x+1) transformed. Differences between otoliths from the 2 coastal calibration sites were tested using both univariate (analysis of variance; ANOVA) and multivariate (multivariate analysis of variance; MANOVA) techniques. For MANOVAs, Pillai's trace was used as the test statistic as it is robust, especially when variance-covariance matrices are not similar (Quinn and Keough 2002).

ANOVAs were performed for each elemental ratio. A Box's M-test was used to determine the equality of variance-covariance matrices and a quadratic discriminant function analysis (DFA) and jackknife cross-validation procedure were used to evaluate how accurately otoliths could be assigned to coast. Finally, otoliths from adult common snook collected from Florida Bay were applied as the test data set to a DFA using otoliths from Tequesta and Charlotte Harbor as the calibration data set to determine to which coastal group the adults were assigned. We acknowledge that this method (DFA) creates a best case scenario and have considered this in our interpretation.

## RESULTS

Three of the 6 elemental ratios of otoliths from the 2 coastal locations differed significantly (Table 1; ANOVA; Mn/Ca: $F_{1,38} = 39.67$, $P < 0.05$; Sr/Ca: $F_{1,38} = 126.41$, $P < 0.05$; Ba/Ca: $F_{1,38} = 114.37$, $P < 0.05$) and MANOVA indicated a significant difference in the multi-element signatures of the juvenile otoliths ($F_{12,106} = 18.17$, $P < 0.0001$). In addition, a DFA depicted a clear separation between the coastal groups and otoliths were classified to coast with 100% accuracy by a cross-validation procedure (Figure 2). A DFA using the juvenile otoliths as a training data set and the adult cores as the test data set indicated
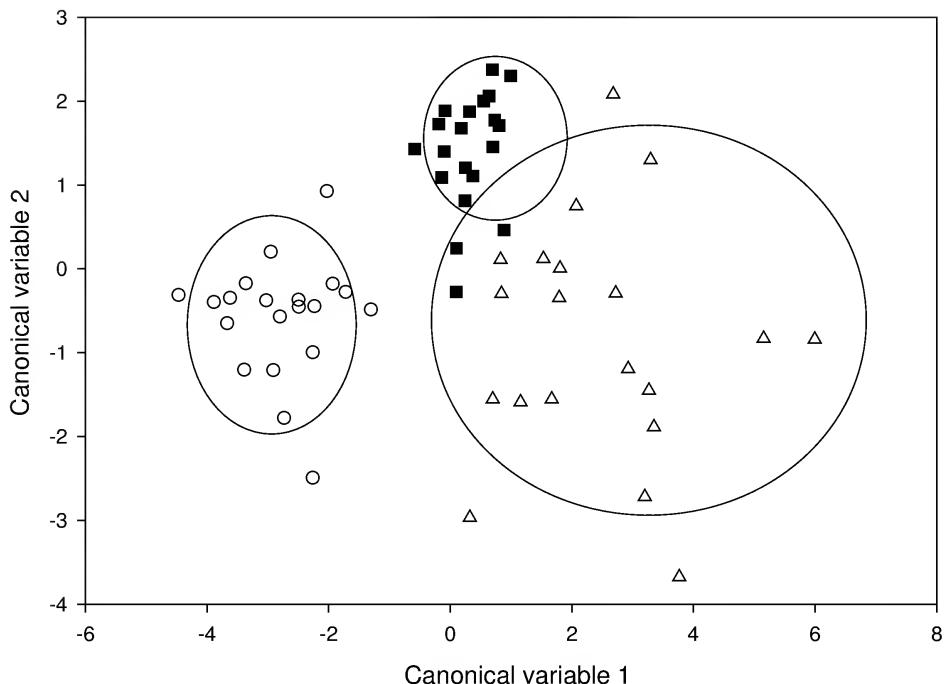
**Figure 2. Canonical plot scores and 95% confidence ellipses from the discriminant analysis of multi-element signatures (Mg/Ca, Mn/Ca, Zn/Ca, Cu/Ca, Sr/Ca, and Ba/Ca) of common snook (*Centropomus undecimalis*) otoliths from Charlotte Harbor (circles), Tequesta (black squares), and Florida Bay (triangles).**

that 45% and 55% of the adult cores from Florida Bay were classified as Atlantic and Gulf coasts, respectively.

## DISCUSSION

The geochemical signatures in the otoliths of juvenile common snook collected from the Gulf and Atlantic coasts of Florida were distinct. This difference in otolith chemistry presumably mainly reflects the differences in water chemistry for each coast, as well as the distinct terrestrial inputs for each estuary (Bath et al. 2000). It was not possible to match the juveniles and adults by year class. Although temporal variation of otolith chemistry within a location has been demonstrated in previous studies (Patterson et al. 1999, Gillanders 2002), it seems likely that overall differences in large water masses such as the Atlantic and Gulf would be temporally persistent to some degree. Indeed, elemental signatures of Gulf red drum (*Sciaenops ocellatus*) from several different years (1982, 1985 and 1998) were quite distinct from those of Atlantic red drum (1998 and 1999), suggesting the consistent separation of these water masses and the otolith signatures produced by them (Patterson et al. 2004).

We were not expecting to match the adults to estuary of origin as this would require that all potential source estuaries be characterized, a task clearly beyond the scope

of this study. Instead, the results presented here are limited to identifying the coastal origin of common snook in Florida Bay and suggest that both the Atlantic and Gulf coastal areas contributed in nearly equal proportions to the adult common snook we examined. Extrapolating beyond our data set to make predictions about the relative contribution of each coastal population to the entire Florida Bay assemblage is not prudent at this time given the limited spatial coverage in our calibration data set. However, this preliminary finding does support the idea that both populations contribute to the Florida Bay common snook assemblage.

These geochemical results concur with those obtained from a genetic study that demonstrated common snook in western Florida Bay exhibited transitional properties of both Atlantic and Gulf coast stocks, and thus both stocks likely contributed to the Florida Bay assemblage (Tringali and Bert 1996). However, the required type of genetic markers (i.e., microsatellites) and likelihood-based statistical methods for assigning individuals to genetically subdivided stocks (e.g., Wasser and Strobeck 1998) postdated their study, so relative contributions of Atlantic and Gulf populations could not be estimated.

In contrast, the geochemical and genetic results do not readily agree with the available tagging data demonstrating that tagged common snook from east coast, but not west

coast locations have moved into Florida Bay. Of the 19,410 common snook tagged on the east coast during 1984–1997, 2 were recaptured inside Florida Bay (Bruger and Whittington, unpublished data). In contrast, of the 8,655 common snook tagged on the west coast during 1976–1986, none were reported as recaptured in Florida Bay (Bruger and Whittington, unpublished data). However, the recapture ratios for each coast were not significantly different. These tagging studies were, therefore, inconclusive regarding the origin of common snook in Florida Bay.

Our results derived from the geochemical signatures of common snook otoliths suggest that both Atlantic and Gulf coast populations in Florida contribute to the common snook assemblage found in Florida Bay. The east-west common stock boundary for management of common snook in Florida occurs at Jewfish Creek in the Upper Keys. This boundary places common snook from Florida Bay and the Florida Keys into the Gulf stock. Common snook occurring north of this line are assigned to the Atlantic stock. The evidence reviewed here suggests the position of this boundary may need to be reevaluated or that the Florida Bay/Keys assemblage may need to be considered separately for management purposes. Future efforts should encompass multiple methods (i.e., genetics, otolith chemistry) and a more detailed spatial analysis of fish from both source areas (east and west coasts) and within Florida Bay to account for the likelihood that both coasts are a source of common snook to parts of Florida Bay.

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# Gulf and Caribbean Research

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## Parasites of the American White Pelican

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# PARASITES OF THE AMERICAN WHITE PELICAN

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*ABSTRACT* Metazoan symbionts, including parasites, infecting the American white pelican (AWP) *Pelecanus ery-throrhynchos* comprise a list of 75 species, 7 of which are new host records. Several new geographic records are also presented, but generally these have a low value because of the migratory nature of the bird. Evidence suggests that some parasites, mostly flies and other arthropods but also nematodes and digeneans, produce detrimental behavioral or pathologic changes in the AWP. Some of the arthropods transmit microbial agents to the pelican. Two digeneans that have the AWP as a definitive host harm and even kill their catfish intermediate host, especially in aquaculture, and another causes abnormalities and mortality in amphibians. Some of the arthropods with low host-specificity can potentially transmit harmful microbial agents to humans and domestic animals. A few avian blood-flukes, intestinal flukes, and nematodes can potentially cause "swimmers itch," gastroenteritis, and "anisakiasis," respectively, in humans. Because of the life cycles of some helminths, presence of those worms can provide an indication as to the dietary items of a specific pelican individual, where the individual has been, and how long it has been present in an area. Feather mites, lice, and diplostomoid digeneans serve as good parasites to indicate phylogenetic relationships among different pelican species as well as relationships among the Pelecanidae and other families such as the Sulidae.

## INTRODUCTION

The American white pelican (AWP) *Pelecanus ery-throrhynchos* serves as a host for numerous parasites, several of them recently or not previously reported. It serves as an acceptable host for many parasites because it migrates over an extensive, defined, geographic range, all the while feeding on a large variety of prey items. Those prey species in turn serve as intermediate hosts for numerous parasites of the AWP, and many of those parasites as well as other symbionts without such complicated life cycles have co-evolved with the pelican. Consequently, knowledge about the symbionts, especially the truly parasitic fauna, provides knowledge about the dynamics of the bird host during certain seasons, in certain regions, or through time. Further, the AWP nests in groups, allowing for easy transference of certain parasites among flock cohorts.

Parasites of birds have been used before by several researchers as indicators of the bird's biology. For example, digeneans (Bartoli 1989) have indicated host diets, prey preference, habitat where prey was eaten, and host migration, even though digeneans and other parasites of fishes have been used more often than those of birds as well as used for additional categories of biological indications (e.g., Kabata 1963, Margolis 1963, MacKenzie 1983, Lester 1990, Overstreet 1993, 1997). These articles cite criteria one should meet, which vary considerably, depending on the type of question the biological indicator should answer. Analysis of the phylogeny of bird parasites as well as the co-evolution or host-switching with their hosts also have concerned lice (e.g., Page et al. 2004) and other parasites. That recent study and others involving

other host groups (e.g., Nadler and Hafner 1993) have strengthened our understanding of phylogeny by taking advantage of a variety of molecular approaches rather than performing strictly morphological analyses. Molecular techniques now provide additional useful tools to investigate a variety of parasitological problems.

This article intends to provide a preliminary foray into many of the AWP's symbionts. "Symbiont" is a general term used here to encompass organisms that cover a complete range of benefit and harm to a "host," whether commonly referred to as a "parasite" or not (Overstreet 1978). They include organisms ranging from those with only a loose bond, such as a stable fly, to those "true" parasites like the digenean *Bolbophorus damnificus*, which is specific to pelicans and exhibits a complicated life cycle in which the intermediate hosts but not the pelican host are harmed. For purposes of this study, we usually refer to each of the symbionts, regardless of the degree of host-specificity, ability to harm a host, or other aspects of its association with the host, as a parasite. Some of these negatively influence the bird, pose a potential public health risk, or provide useful biological information about the pelican or its associates. The focus of this article is not meant to emphasize disease in the AWP resulting from parasites but rather to show how parasites may play an important role in understanding the biology of the bird.

## MATERIALS AND METHODS

Data for this article have been gathered from prior collections made by the authors, specimens borrowed from museums, and the literature. Collections consist of parasites from a relatively small number of the AWP. Those

birds from the Mississippi-Louisiana region were shot or died during their migration in conjunction with the USDA/APHIS/ Wildlife Services, provided by the Wildlife Rehabilitation & Nature Preservation Society, or collected from aquaculture facilities. A single specimen was shot at the Summer Lake Wildlife Area, Oregon. We compare our data on the AWP with that collected or described from the brown pelican (*Pelecanus occidentalis*) in Mississippi or elsewhere. Parasites were removed from birds that were either freshly killed or birds that had been collected earlier and their alimentary tract frozen or fixed in alcohol or formalin. A few birds were examined thoroughly for parasites in general, and specific organs of others were examined for certain species of parasites. Parasites were prepared using a variety of methods common in our laboratory, including some described by Cable (1977), Crites and Overstreet (1991), and Overstreet and Curran (2005). In the most recent collections, we placed representative specimens in refrigerated 95% molecular grade ethanol or directly into a −70 °C freezer for molecular studies.

## RESULTS

Appendix 1 lists most of the parasites about which we have knowledge and provides the site in the bird, the geographic location of the infection, and principal references, when applicable. When we considered recorded parasite names to represent junior synonyms, we usually included the accepted names only. We also include in the table infections of parasites that we encountered for this study. The list, plus a single unidentified coccidian protozoan, contains 75 species, 9 of which are flies and 1 a flea with very little specificity toward the AWP but important to the bird's health. About three of the reports represent synonyms or misidentifications of binomial species already on the list. That leaves about 62 listed species, not counting the flies and flea, even though some unidentified species probably represent a complex of species. Of the 62, about 24 have not been reported from the brown pelican (e.g., Dyer et al. 2002). Many are confined to both pelican species (e.g., Dronen et al. 2003), but others infecting both pelicans also infect other birds. About 10, consisting of lice, mites, and helminths, infect the AWP only. Questionable identifications and an indication of whether infections also occur in the brown pelican are noted in Appendix 1. New geographical records are not individually marked. Appendix 1 also lists seven new records for the AWP based in some cases on careful examination of just a few specimens. These are marked with an asterisk. Even with the inclusion of this study, historically few specimens of the AWP from few localities have been thoroughly

examined for parasites. And from these few individuals, often the site of infection has not been recorded. As a result, we predict that many additional parasites infect the AWP. For example, the brown pelican hosts the eimerian coccidian protozoan *Eimeria pelecani*, but no protozoan other than a record of the coccidian *Sarcocystis* sp. in cysts in the pectoral muscle (Forrester and Spalding 2003) is known yet from the AWP.

## DISCUSSION

Because some parasite identifications were based on single, few, or incomplete specimens and because some identifications or infections required annotation, we mention a few relevant points. For the cestodes, no scolex was found among our tapeworm material of *Paradilepis* cf. *caballeroi*, so the species could not be identified, but, based on the diagnosis by Forrester and Spalding (2002), we assume *P. caballeroi* is probably correct. There exists some debate about the status of species in *Paradilepis* and related genera. *Parvitaenia heardi* (= *Glossocercus caribaensis* by Scholz et al. 2002a) was described from the great blue heron (*Ardea herodias*) in South Carolina by Schmidt and Courtney (1973), with the brown pelican listed as an accidental host. Rysavy and Macko (1971) reported *Parvitaenia eudocimi* (= *Cyclustera ibisae* by Scholz et al. 2002a) from the brown pelican in Cuba as well as from the white ibis (*Eudocimus albus*), for which it was named. They described *P. caballeroi* from the double-crested cormorant (*Phalacrocorax auritus*). Additional well-fixed material of dilepidids from the AWP should allow exact identifications and perhaps reveal the presence of several additional species. For example, Scholz et al. (2002b) found five different dilepidid species in "*Phalacrocorax olivaceus*" (= the neotropic cormorant, *P. brasilianus*) in Mexico.

For the digeneans, *Bolbophorus confusus* has been reported by several authors as one of a few diplostomoideans infecting the brown pelican and AWP. Overstreet et al. (2002) have shown that at least some of those records represent one and in some cases two related species of *Bolbophorus*. Whether *B. confusus*, which we consider to be a European species, exists in North America still has to be determined (Overstreet et al. 2002). One of the two species previously reported in part as *B. confusus* by at least some authors (e.g., reviews by Olson 1966, Overstreet et al. 2002, Appendix 1) was considered as *B. damnificus*, and it can be readily acquired by the AWP from feeding on infected channel catfish (*Ictalurus punctatus*) (Overstreet et al. 2002, Overstreet and Curran 2004). The other species is referred to as *Bolbophorus* sp. of

Overstreet et al. (2002). It has been demonstrated to differ from *B. damnificus* as shown by sequences of four different gene fragments by Overstreet et al. (2002) and corroborated by Levy et al. (2002) using one fragment. Levy et al. (2002) showed that it infected several fishes but not the catfish. Adults of both of the two species can occur in the same individual pelican host (Overstreet et al. 2002).

When we sequenced the ITS 1/2 for three preadult specimens from a single AWP in Oregon, the percentage value for DNA sequence similarity with *Bolbophorus* sp. of Overstreet et al. (2002) was 99% in contrast with 88% when compared with the same fragment of *B. damnificus*. The three specimens also had a similarity value of 99%, indicating the immature Oregon specimens were *Bolbophorus* sp. of Overstreet et al. (2002). Also, based on specimens of diplostomoideans we have seen from northern North America, we suspect that the report of *Diplostomum spathaceum* from the AWP in Manitoba by McLaughlin (1974) probably represents a species of *Bolbophorus*. He found only three helminths, and only D. spathaceum was a digenean. Since it appears superficially similar to the two species of *Bolbophorus* that are common in the AWP in Canada, we treat the report as a misidentification.

We found another diplostomoid reported as *Bursacetabulus pelecanus* in the AWP as well as in the brown pelican. Whether it is conspecific with *Bursatintinnabulus macrobursus*, which we also found in both local pelican species, is being treated by Charles Blend, Overstreet, and Curran (unpublished data).

Several host records for digeneans deserve comment. *Ribeiroia ondatrae* has been reported from various gulls and the muskrat. McNeil (1949) listed the AWP as a host in Washington, Forrester and Spalding (2002) reported it from Florida, and Dyer et al. (2002) reported it from the brown pelican in Puerto Rico. Originally, Price (1931) did not notice the esophageal diverticula and considered the species in a different genus, but Lumsden and Zischke (1963) confirmed their presence in the type material. The presence of this worm in pelicans is important because of the effect of the species on its amphibian intermediate hosts discussed below. Species of *Renicola* are difficult to differentiate, primarily because eggs obstruct the view of most of the organs. We appear to have two species. Specimens from Mississippi are similar to *Renicola thapari*, but our relatively young specimens appear different enough from the much larger and more fecund specimens described from the brown pelican in Panama (Caballero 1953) and reported later from that host in Florida and Louisiana by Courtney and Forrester (1974) to consider the identification tentative. A portion of a damaged speci-

men collected from Oregon seems to represent a different and possibly new species or one of the few renicolids that infect other white pelican species (e.g., Stunkard 1964). The identification of *Prosthogonimus ovatus* was based on a single specimen from the wash of the oviduct and a small portion of the cloaca, but it was not initially observed in the cloaca. It has a smaller body, suckers, and eggs than *Prosthogonimus folliculus* from the American bittern (*Botaurus lentiginosus*), and, based on reported North American species of the digenean, it seems most consistent with the description of *P. ovatus*, a species known from several birds, both small and large. Actually, we expect specimens that have been reported from different cosmopolitan hosts as *P. ovatus* to represent a complex of species. Our measurements of what we identified as *Austrobilharzia variglandis* are slightly smaller than those reported by Stunkard and Hinchliffe (1952) and may represent an atypical infection. In any event, the eastern mudsnail (*Nassarius obsoletus*) (also known as the mud whelk, *Ilyanassa obsoleta*), intermediate snail host for *A. variglandis*, occurs along the northern Gulf of Mexico as well as along the eastern US seaboard, where infections in it have been investigated (Barber and Caira 1995). Gulls appear to be the primary avian host for the species. *Gigantobilarzia huttoni* (see Leigh 1957) and *Dendritobilharzia pulverulenta* (see Forrester and Spalding 2003) infect the AWP in Florida and presumably elsewhere, and other blood flukes infect the brown pelican and other pelicans around the world (e.g., Yamaguti 1971).

Nematodes in the genus *Contracaecum* require a taxonomic revision. There have been six nominal species reported from the AWP, but, considering a synonym and misidentifications, we considered only four species acceptable; occasionally at least three species occur concurrently in an individual bird. The morphological features of the species do not fit all the descriptions corresponding to the names (e.g., Deardorff and Overstreet 1980). In any event, we have seen three species listed in Appendix 1 as concurrent in the AWP from Mississippi and Louisiana, with *C. multipapillatum* being the most common in those localities. As the populations of both the AWP and the brown pelican increase, the juvenile infections in the two local mullets (*Mugil cephalus* and *Mugil curema*, common second intermediate hosts) become more abundant, resulting in pelican infections, which in turn commonly reach over several hundred specimens in an individual bird. In Oregon, we have seen *C. microcephalum* as identified using the work of Baruπ et al. (1978), and only several specimens occurred in the bird. What we call *C. microcephalum* in North America may be a distinct species but closely related to the European form. *Contracaecum*

*microcephalum* has been reported throughout the world. Heavy infections by a species reported as *C. microcephalum* from Tanzania were held responsible for pelican mortality there (Nyange et al. 1983).

The external features of the clitellate glossiphoniid leech *Theromyzon* sp. from Oregon differed from those described for *T. rude*, but the species is clearly in the genus. Six young specimens were restricted to the breast, neck, and head area, and none was associated with the cloaca. Mark Siddall and Elizabeth Borda (American Museum of Natural History) are in the process of sequencing material to establish its identity. No leech has been reported previously from the AWP, but Rothschild and Clay (1957) mentioned that leeches occurred in the vent and gular pouch of pelicans.

Feather mites are treated below under pylogenetic relationships. We expect several more feather mites infest the AWP than have been reported. For example, nymphs of three species of hypoderatid mites in the subcutaneous tissues of the brown pelican were reported in Louisiana and Florida by Pence and Courtney (1973), and we have seen unidentified, presently unavailable for study, species in the subcutaneous fat around the trachea of the AWP. Adults of these mites inhabit the nests. Also, the trombiculid *Womersia strandtmanni* has been reported in the brown pelican by Vercammen-Grandjean and Kobebinova (1968). That chigger caused skin lesions in ducks (Clark and Stotts 1960). Based on knowledge of mite infestations in other pelican species, we suspect related or identical species occur on the AWP.

## Bird health

Depending on what one wishes to consider a cause of disease, there could be several of the organisms listed in Appendix 1 that have a direct or indirect negative influence on the health of the AWP. The fleas, ticks, and flies all can pose a threat to the health of the bird, especially weakened young, captured, or disabled individuals. These have low host-specificity with the pelican; for example, there are eight species of flies listed and presumably many more exist. These arthropods have been observed on young individuals in "nesting areas" in large numbers, and often the birds in question died (Johnson 1976). Whether the young were unhealthy and attracted the flies or whether the flies caused the birds to become unhealthy is uncertain, but, in any event, the flies aided in the demise of many individuals. Johnson (1976) found the adult flies annoying young birds that hatched primarily late in the season in Chase Lake National Wildlife Refuge. The young birds were unable to avoid the flies by moving into open areas, and, once a few flies started feeding on a bird's flesh in the head or else-

where, many more became attracted to feed, and these laid eggs in the bird, resulting in even greater numbers.

The flies offer additional means of causing disease. Because they are not specific to the pelican, they often leave one individual or one host species and find another. Even though hippoboscid flies have little ability to fly and infest their hosts primarily through direct contact, most other flies like the blowflies and stable flies travel from host to host. Consequently, those that fly the farthest can more readily pick up a bacterial, viral, or some other infection from one wild or domestic host species and transmit it to another host such as the AWP or even a human.

The soft tick *Ornithodoros capensis* presents another problem. This common argasid has been held responsible for causing the parent birds to desert their nests, sometimes for two years. As indicated above, they are not specific to the AWP, but they infect several different aquatic birds. King et al. (1977a) found that three deserted brown pelican nests in Aransas National Wildlife Refuge, Texas, yielded 2,389 adult and nymphal specimens of the tick. This and perhaps another species (*Ornithodoros denmarki*) probably caused nest desertion by the brown pelican in Gulf of California nests. Scratching and preening behavior occurred from 32–68% of the morning and afternoon observation times in areas where desertion was greatest (King et al. 1977b). Death of the nestlings may result from transmission of a lethal Soldado-like arbovirus from the tick (Converse et al. 1975). Infestations also are known to reduce brood size in Texas (King et al. 1977a, 1977b). We think the actual importance of this tick to the AWP probably depends on air temperature. Infestations have been reported on the AWP in Texas (King et al. in Duffy 1983), where temperatures remain relatively high. We questioned various biologists such as Robert Johnson and Kory Richardson at Chase Lake National Wildlife Refuge, North Dakota, and Marty St Louis at the region in and near Summer Lake Wildlife Area, Oregon, and northern California, and they never recalled seeing any ticks on the birds or in their nests from these relatively cool nesting grounds.

Perhaps other agents also cause pelicans to desert their nests. Rothschild and Clay (1957) mentioned that entire colonies of pelicans in the southern seas have deserted their nests because of *Culex pipiens*, referred to as a "house-gnat" rather than a mosquito. The complex of mosquito species in the *C. pipiens*-group has been held responsible for extinction or shifting ranges of various bird populations because it transmitted both bird malaria and avian pox virus (e.g., Warner 1968). We do not include the non-specific mosquitos in Appendix 1.

Lice are much more specific to the pelican than are the flies; in fact some lice species apparently infect no other bird except the AWP. They feed on blood and can occur in the thousands on birds that cannot adequately preen themselves, such as weakened young, captured, and disabled individuals. When someone encounters a pelican with large numbers of lice, the person should assume that the individual bird is in poor health. On one specimen of a brown pelican from Mississippi in September 1993 with a distorted bill, we observed thousands of specimens of lice, primarily of *Pectinopygus occidentalis* but some of *Colpocephalum occidentalis*, on its head, back, and wing feathers. The same occurs with the counterparts *P. tordoffi* and *C. unciferum* on the AWP. In fact, we observed feather mites associated with the lice on the brown pelican. The "pouch louse," *Piagetiella peralis*, is a biting louse that cannot be controlled by normal preening because it occurs in the gular pouch. The healthy AWP usually keeps an infestation in check, but weakened individuals often exhibit hemorrhagic ulcerative stomatitis and inflammation of the mouth (Wobeser et al. 1974, personal observations). The effect may be serious, and infestations are readily transmitted to young during nesting when infested parents are feeding them. Not all individuals of *P. peralis* infest the inner surface of the gular pouch, where its large numbers can cover the entire surface along the lower mandible and on the roof of the mouth without producing severe lesions. On 250 examined young birds, Johnson (1976) found them in the pouch of all, and 53 of a subset of 90 had some at the base of the neck, bottom of the feet, and axil of the wings. He noted that such external infestations appeared to subside after the birds reached 2–3 weeks of age. Wobeser et al. (1974) reported a large number of immature lice over the entire body of a dying young juvenile. In the only examined adult from Oregon, we found, in addition to those in the pouch, numerous immature specimens tightly lodged along the shaft of the primary wing feathers and a few younger specimens among the breast feathers. None was associated with pathological alterations.

Whether helminth infections harm the AWP depends on factors such as the number of worms present, prior state of the bird's health, and bird age. Individuals of some members of the anisakid nematode genus *Contracaecum* often occur in the hundreds in the proventriculus and adjacent organs of the AWP. Oglesby (1960) estimated over 1,100 individuals from a single AWP that had died in Florida. These were tentatively identified as *C. micropapillatum*, a species that Deardorff and Overstreet (1980) found in low numbers concurrent with considerably larger numbers of *C. multipapillatum* and *Contracaecum rudolphii* in other specimens in Mississippi and Louisiana.

Adult and fourth stage individuals typically associate with an ulcer where they attach and perhaps feed on the host response tissue. The secretions and excretions by juvenile worms are probably more responsible for local inflammation and necrosis than those by adults (Liu and Edward 1971, Fagerholm et al. 1996). We have seen ulcers both with a well-delimited conspicuous fibrotic protective capsule, allowing the nematodes to feed on inflammatory cells without disturbing the adjacent stomach tissue, and without such encapsulation. When without the capsule, the lesion is typically associated with extensive inflammation. After the bird host feeds, the nematodes often detach from the ulcer and entwine among the prey material. When individuals of various species of *Contracaecum* were found present in large numbers, some observers (Owre 1962, Huizinga 1971, Fagerholm et al. 1996) suggested that they help macerate or digest the prey as an initial stage in the host's digestive process. *Contracaecum multipapillatum* and related species also have been suggested as being associated with mortality or poor health of the bird host. Morbidity of hosts of all ages can be suspected when at least some individuals of a relatively large worm burden penetrate through the mucosal layer, when infections have an associated secondary microbial infection, or when an individual is starved (e.g., Oglesby 1960, Owre 1962, Fagerholm et al. 1996). Dyer et al. (2002) also suggested that the same species may have contributed to the emaciation and death of brown pelicans in Puerto Rico. Grimes et al. (1989), who tested the effects of four anthelmintics on *Contracaecum* spp. and two digenean species in the brown pelican, mentioned unpublished data by Courtney, who demonstrated that nestling pelicans with 95% of the nematodes removed by treatment showed higher weight gains than untreated controls.

Digeneans can also harm the AWP. As with some of the other agents indicated above, the pathological effect often depends on the number of individuals and other factors such as a secondary bacterial infection. *Phagicola longus*, a small species, probably affects the AWP that nests along the coast because a marine snail and mullets act as intermediate hosts. For example, nestling brown pelicans 4–5 weeks old from Louisiana contained over 18,000 specimens of *P. longus* along the small intestine and ceca, many in the mucosa and lamina propria. *Mesostephanus appendiculatoides*, present in lower numbers (e.g., averaging 1,112 specimens per bird from the Floridian Gulf of Mexico coast) attached to the villar tips and occasionally penetrated the epithelium (Humphrey et al. 1978, Greve et al. 1987). It was acquired from coastal silversides as well as mullets. Both digeneans, also occurring in the AWP, distorted host tissues and produced an inflammatory response

but did not kill birds in captivity. As few as 15 specimens of the larger *Ribeiroia ondatrae* deep in the proventricular mucosa produced necrosis, possibly contributing to mortality of the brown pelican (Dyer et al. 2002). According to Rebecca Cole (personal communication, National Wildlife Health Center, Madison, Wisconsin), a heavy infection of *Pholeter anterouterus* along the intestine of an AWP in Florida killed the bird, possibly in conjunction with an acid fast bacterial infection.

The diet of fish allows the potential for harmful effects in addition to helminth infections. Because fish bio-accumulate various pesticides and other toxic agents, pelicans and other piscivorous birds can further accumulate such compounds (e.g., Forrester and Spalding 2003). Well-documented cases of the brown pelican with bioaccumulation of high levels of DDT and other pesticides in the late 1950's and early 1960's and then other pesticides in 1975 resulted in thin eggs and loss of fledged offspring. The reduced production of young decimated the brown pelican population in the northern Gulf of Mexico and other areas, and a return of successful breeding colonies took several years (Johnsgard 1993). A condition of far less concern involves older individuals feeding on physically dangerous items. Lesions commonly observed by us in the stomach suggest that punctures by spines such as those on pectoral and dorsal fins of catfish and other prey can develop secondary infections and perhaps produce death when the prey is not eaten head-first. Catfish spines killed two adult AWP (Forrester and Spalding 2003); one lacerated the jugular vein and the other perforated the esophagus and stomach. Johnson (1976) observed two young birds, one of which died, with penetrating fish vertebrae lodged in their throat. Related to this kind of damage was a case of potential death resulting when an AWP engulfed a wooden- handled ice-pick (Mattis and Deardorff 1988). The bird with the pick had difficulty standing, remained in a squatting position with a contracted neck, could not fly, and could no longer feed or be force-fed. Once the bird was x-rayed and the condition diagnosed, the pick was shown to have entered down the esophagus handle first and perforated that organ, so that the pick could be removed from its lodgment and the bird saved only by human intervention.

**Health of intermediate hosts**

Not only can the AWP be harmed by a few species of helminth parasites, but a few of the helminths that have little effect on the pelican can be transmitted by the pelican and have a drastic influence on the intermediate host population. Good examples include two diplostomatid digeneans that infect catfish and a cathaemasiid digenean that infects amphibians. These diplostomatids, *Bolbophorus*

*damnificus* and *Bursacetabulus pelecanus*, both can produce mass mortalities of the channel catfish, at least in aquaculture conditions (Overstreet et al. 2002, Overstreet and Curran 2004). The problem with *B. damnificus* is more confusing than originally presumed by fish farmers and managers because more than one species of *Bolbophorus* infects the AWP (Overstreet et al. 2002), with a single individual bird capable of harboring at least two of those species. Only one of these is known to infect the catfish. For the two indicated species that infect the catfish, their eggs are released with the pelican's feces into the aquaculture ponds. The miracidia (infective larvae) of both infect the appropriate snail host, and, after development of at least two asexual stages of the digeneans and ultimate production of large numbers of infective cercariae, individuals of the cercaria of each species are shed in large pulses available to infect the catfish. Those for *B. damnificus* enter the fish and finally lodge and encyst, typically in the muscle adjacent to the dermis in the caudal region, and those of *B. pelecanus* end up unencysted in the vitreous humor of the eye. Infection by *B. damnificus* also results in pathological alterations in the kidneys (Overstreet and Curran 2004). The snails and up to millions of associated cercariae occur along the shallow sides of fish-ponds where young catfish occur and receive massive infections, often resulting in death (e.g., Terhune et al. 2002). We have exposed catfish to the cercariae of *B. damnificus* in the laboratory and produced death of the fish after periods ranging from minutes to days, depending on the dose of cercariae (Overstreet et al. 2002, Overstreet and Curran 2004). How many cercariae of *B. pelecanus* are necessary to harm the catfish was not established, but infections of another diplostome, *Austrodiplostomum compactum*, which matures in various cormorant species, infected the vitreous humor as well as the brain and spinal cord and also killed the catfish. For it to kill the host necessitated a larger number of the penetrating cercaria than did *B. damnificus* in short-term laboratory infections. Thousands of very young worms could infect the nerve tissue of the fish (Overstreet and Curran 2004). Consequently, the AWP does not necessarily serve as the only avian source of digeneans that can cause catfish mortalities and it is not the only scourge of the fish farmers wanting to rid their ponds of pelicans. The AWP and different cormorants eat catfish from the ponds, whether the catfish are infected or not. The example involving harm to amphibians concerns the cathamaesiid digenean *Ribeiroia ondatrae*. The metacercaria of this species produced limb malformations in a wide range of amphibians (frogs, toads, newts, and salamander) in wild and experimental hosts (Johnson et al. 2002), with survivorship declining significantly with

increasing cercarial exposure (Johnson et al. 2001). Unlike the examples of diplostomids where the pelicans and catfish are the only known hosts, *R. ondatrae* infects a few different vertebrate definitive hosts in addition to pelicans as well as numerous amphibian second intermediate hosts and several snail species of the planorbid genus *Planorbella* as first intermediate hosts.

A similar problem involving harm to the intermediate host concerned recreational fishermen and those interested in the AWP from the late 1920s until many years after in Yellowstone Lake, Wyoming. The pelican colony on Molly Island had to be protected because the birds there transmitted the tapeworm *Diphyllobothrium cordiceps* to the local trouts. When Behle (1958) wrote on the AWP, he indicted that the Park Service officials then felt that the value of the birds offset the loss of available fish.

**Health of humans and domestic animals**

As indicated above, some of the flies can transmit microbial infections to pelicans and other hosts, including humans. For example, the blood-feeding stable fly *Stomoxys calcitrans*, as summarized by Roberts and Janovy (2000), can transmit the flagellates *Trypanosoma evansi* and members of the *Trypanosoma brucei*-complex, the agents of surra and sleeping sickness in large mammals, as well as epidemic relapsing fever, anthrax, brucellosis, swine erysipelas, equine swamp fever, African horse sickness, and fowl pox. This species also serves as the intermediate host for the nematode *Habronema microstoma* in horses. These infections are in addition to the biting that causes severe discomfort in humans and death in livestock. Some of the other flies can also transmit various agents. The possibility of the lice transmitting an agent has not been investigated, but that by pelican-ticks has been studied minimally (Forrester and Spalding 2003). One can say in general that transmission of numerous avian viruses, many of which are transmitted by arthropods, can have a serious negative influence on domesticated and wild birds and mammals as well as on humans (Perdue and Seal 2000).

The argasid soft tick *Ornithodoros amblus*, which acquires short blood meals off the brown pelican and other seabirds in nesting islands off Peru, has been associated with the birds deserting their eggs and young. It possibly transmits infectious agents to the birds. At least two arboviruses, "Huacho" and Salinas," are transmitted by the tick. Although the effect on the birds was not established, Duffy (1983) reported that humans suffered swelling, itching, occasional gangrene, and even death following multiple tick bites.

Helminths also can be spread by the AWP to humans. For example, when the bird infected by the schistosome *Austrobilharzia variglandis* defecates in marine waters containing the eastern mudsnail (*Nassarius obsoletus*), the snail can get infected by the miracidia (larval stage hatched from the worm's egg) and this larva undergoes asexual reproduction, ultimately producing many thousands of cercariae. The cercaria is the invasive stage shed from the snail that infects the AWP or a variety of gulls and shore birds (e.g., Barber and Caira 1995). If it invades a human rather than the bird, it does not develop, but rather it establishes a host sensitivity response such that future invasions result in a hypersensitivity reaction in the skin of one who inhabits water containing the cercaria. As the host's defense responses react against the challenging doses of the cercaria, allergins are released from the cercaria that cause an inflammation. This reaction, called "swimmer's itch" or "clam digger's itch," is painful enough to keep people from entering beaches and other bodies of water that contain infected snails; and, consequently, public swimming areas often are closed, producing a local economic hardship. Patients are seldom severely harmed, but the hypersensitivity reaction keeps most from revisiting the location. Unlike the two-host schistosome life cycle, most helminths utilize a series of at least two intermediate hosts plus the definitive host. For example (e.g., Huizinga 1967), when a bird with the nematode *C. multipapillatum* defecates in near shore or freshwater habitats, some cyclopoid and presumably other copepod species feed on the released larval nematode, supporting development to a stage (third stage) or condition (exsheathed second stage) infective to a fish intermediate host. The larva, or more appropriately the "juvenile," can develop only in certain fish species. When other fish, or in some cases invertebrates, are eaten by animals other than the AWP or other avian definitive hosts, the worm migrates to the body cavity, becomes encapsulated, and remains infective to a pelican that feeds on the animal. Small fish intermediate hosts can be killed by the worm. Our original research based on non-human animals suggested that if humans ate this fish (e.g., primarily the striped mullet but also the red drum, *Sciaenops ocellata*, and other fishes), the worm would be digested (Deardorff and Overstreet 1980). However, later research involving RMO (Vidal-Martínez et al. 1994) showed that in some cases, presumably involving a warm period of acclimation, the worm could produce "anisakiasis." The term "anisakiasis" defines a disease in warm blooded mammals including humans caused by various ascaridoid species in the family Anisakidae and not just those in the genus *Anisakis*. Because of the recent increase in brown and white pelican infections following reduced

levels of DDT and related compounds, the potential risk is increasing. During that same period of depletion and recovery, the striped mullet in the northern Gulf of Mexico was overfished for the Japanese caviar industry and other needs. The reduced numbers of both avian and fish hosts subsequently reduced infections of *C. multipapillatum* in mullets, pelicans, and cormorants, but with increases in those hosts, heavy infections are recently beginning to return (unpublished observations).

A public health risk also occurs for those eating inadequately cooked American species of mullet infected with *Phagicola longus*, a digenean infecting a snail that feeds on eggs shed with host feces by the AWP or a few other birds; the fish becomes infected from the cercaria shed from the snail (Overstreet 1978, Deardorff and Overstreet 1991). Unlike the nematode that infects warm-blooded hosts as a juvenile, *P. longus* matures in the warm-blooded host, often causing grossly appearing gastroenteritis in herons and raccoon hosts (Overstreet 1978, Richard Heard and Overstreet, personal observations).

### Indicators of biological activities

As described in abbreviated detail above, helminths undergo a complex life cycle involving two or more different hosts. The AWP is the final, or definitive, host for those listed in Appendix 1. The cycle in different helminth groups differs, and that for each species differs from all others in some ways, usually by the specific hosts involved. Knowledge of these life cycles and life history patterns can provide important biological information on feeding habits and migratory patterns of the host individuals. For example, *Phagicola longus*, *Mesostephanus appendiculatoides*, and *Contracaecum multipapillatum* all infect the striped mullet, *Mugil cephalus*, as the second intermediate host. When these parasites are observed in a pelican, one knows that the pelican has been feeding on mullet along the coast. In contrast, *B. damnificus* and *B. pelecanus* infect the channel catfish in fresh water, usually far from the coast, and the presence of one or the combination of both in the pelican indicates that it was feeding on the catfish. Of course there exists a variety of other helminths from both habitats, but most do not occur in large numbers. By looking at the relative numbers of these freshwater and coastal parasites as well as the presence or absence of each species, one can get a good indication of where the bird has been and how long ago the bird was feeding on what and in what habitat. Since there is a loss of individuals with time, there is a greater likelihood of a recent infection if there is a heavy infection of a species that can occur in large numbers (e.g., those species indicted above). Moreover, this indication of a recent infection

can be strengthened when some individuals of certain species possess few or no eggs, indicative of recent acquisitions. If specimens of parasites from coastal and inland habitats are both present, evaluation of all these features should provide the necessary feeding and migratory information. Humphrey et al. (1978) treated the differences in community structure of the above helminths in the brown pelican from the east and west coasts of Florida and from Louisiana. They pointed out the eventual decline in *P. longus* in adult brown pelicans could result from a possible immune response established during a tissue dwelling stage occurring in the fledgling pelicans. They also speculated on the community structure being influenced by a change from mullet as a dietary item when young birds no longer depended on food from their parents. Kinsella et al. (2004), who collected helminths from the AWP in Florida, noted that most of the helminths from their 29 birds had been acquired in the marine habitat, even though many of the birds were collected inland. The community of helminths in the AWP would probably provide a good model to demonstrate an interactive community (Holmes and Price 1986), especially since the parasites have such a diverse array of effects on the bird populations.

Even though few birds were examined critically by either Dronen et al. (2003) or us and little can be determined from incomplete data on prevalence or intensity, we can surmise that the endohelminths from the AWPs from the Mississippi-Louisiana region had a greater richness than in the counterparts from the Galveston Bay, Texas, area. Ten endohelminths reported from six AWPs from Texas compared with 19, or at least 20 considering synonyms, from Mississippi/Louisiana and with 33 from Florida, where the sample size was much larger (Forrester and Spalding 2002, Kinsella et al. 2004). In all cases, the worms were derived from a combination of freshwater and marine intermediate hosts. In Texas, the brown pelican, a bird that has a more restricted home range than the AWP, had 23 species, a number comparable with those we observed in the AWP but still considerably less than the number of endohelminths that occur in the brown pelican from Florida and presumably Mississippi.

We are also interested in knowing what parasites are residents in intermediate hosts in specific habitats. Specific intermediate hosts and cycles for many helminths have not been discovered. The presence of preadult specimens of *Bolbophorus* sp. of Overstreet et al. 2002 in an AWP on the Summer Lake Wildlife Area, Oregon, suggests that the bird acquired the infection in or near the Area.

## Indicators of phylogenetic relationships

Different tools have been used to discern phylogenetic relationships within and among avian families, including pelicans. For example, Cracraft (1985) presented a closer morphological relationship, based on an extensive cladistical analysis, between Pelecanidae and Sulidae (gannets and boobies) than between pelecanids and the cormorants or anhingas, once thought to be more closely related to the pelecanids than any other birds. But Warheit et al. (1989), using just the number of ossicles per ring in the sclera of the eye's corneal hemisphere, separated the pelicans farther from the sulids than the other bird groups. Then, first using DNA-DNA hybridization (e.g., Sibley et al. 1988) and later using DNA sequences of mitochondrial 12S and 16S rRNA genes (1.7 kb) (Hedges and Sibley 1994), the biologists also separated those groups similarly to the arrangement of Warheit et al. (1989). Siegel-Causey (1997) concluded, as did Sibley and colleagues, that the originally designated Pelecaniformes was paraphyletic (having more than one unrelated ancestor), with none of the several studies supporting a monophyletic (single original ancestor) origin of the order. Only the relationship between the pelicans and shoebills appears consistent with all the molecular data. The author also considered the molecular studies in an elementary stage, with answers requiring a re-examination of traditional morphological characters. Nevertheless, preliminary parasitic data on infections with closely related species of feather mites, diplostome digeneans (species of *Bursacetabulus* and *Bursatintinnabulus*), and cyathocotylid digeneans (species of *Mesostephanus*) seem to support a close relationship of the pelicans with the sulids.

Appendix 1 indicates that many of the AWP parasites also occur in or on the brown pelican. There are a few groups of ectoparasites such as feather mites and lice and endoparasites such as diplostome digeneans and tetrabothriid cestodes that contain counterparts that differ between the two North American pelicans. These and related parasites allow us a better insight into the phylogenetic relationship between the two pelican species as well as among all pelicans and among the Pelecanidae and other bird groups.

Feather mites have been demonstrated to be good parasitic tools to indicate relationships within and among bird groups (e.g., Mironov 1999). For example, members of the genus *Scutomegnina* (Avenzoariidae) on the Pelecanidae and Sulidae show a closer relationship among each other than those from birds of either family show to the mites on cormorants and anhingas (Mironov 2000). Moreover, Mironov (personal communication, Zoological Institute, Russian Academy of Sciences, St. Petersburg, Russia) considers specific mites from the AWP in general more similar to those on other white pelicans than to those on the local North American brown pelican. For example, *Scutomegninia gaudi*, originally described from *Pelecanus onocrotalus*, also occurs on the AWP, but *Scutomegninia remipes* occurs on all the "subspecies" of the brown pelican (Mironov 2000), and *Megalloptes major* occurs on the brown pelican (Mironov and Pérez 2000). We expect *Megalloptes triphyllurus* will be found on the AWP, since it occurs on other white pelican species. *Alloptellus pelecanus*, already known from *Pelecanus onocrotalus*, *P. crispus*, and *P. rufescens* (see Peterson and Atyeo 1972), probably also occurs on the AWP. We predict a species of *Plicatalloptes* to be found on the brown pelican that is different from *P. pelecani*. Presumably, several more mite species will be discovered on the AWP. Because of the large number of named and presumably unnamed feather mites showing various degrees of host-specificity in members of Pelecaniformes, this group of parasites seems the perfect group with which to assess phylogenetic relationships among the birds.

Since feather lice—like feather mites—are different among brown, white, and other pelicans that have been studied, we predict this group will also provide a powerful insight into the phylogeny of members of the genus *Pelecanus*. For example, in North American hosts, *Colpocephalum unciferum*, *Pectinopygus tordoffi*, and *Piagetiella peralis* infest the AWP in contrast with *Colpocephalum occidentalis*, *Pectinopygus occidentalis*, and *Piagetiella busaepelecani*, which occur on the brown pelican. Initially (Kellogg 1896), *C. unciferum* was thought to infest both pelicans, but it was later shown to be different from the material on the brown pelican. Additional related species infest other pelican species and other related species. A cladistical analysis of the species should reflect phylogenetic relationships among all pelecaniforms, including the ancestral association among the different pelicans. Of the Pelecaniformes, pelicans and frigate birds are infested by members of *Colpocephalum*, but birds in several other orders are also infested (Emerson 1972). Members of *Pectinopygus* infest some birds in every pelecaniform family except Phaethontidae (tropicbirds), with several species on pelicans, boobies, and gannets as well as frigate birds, cormorants, and anhingas. Members of *Piagetiella* infest only pelicans and cormorants (Price 1970).

Members of the diplostomoid digenean genus *Bursacetabulus* are known from pelicans and a gannet only. *Bursacetabulus pelecanus* infects the brown pelican (Dronen et al. 1999) and the AWP, and *Bursacetabulus morus* infects the northern gannet (*Morus bassanus*).

Additionally, two other nominal species in the genus *Bursatintinnabulus* are reported from the same hosts (Tehrany et al. 1999), although we question the taxonomic status of those latter worms.

Six genera of cestodes in the family Tetrabothriidae have shown the genus *Tetrabothrius* to be pleisiomorphic (=ancestral) (Hoberg 1989, Hoberg et al. 1997). Members of the genus suggest an archaic association of the species among the Pelecaniformes, Procellariiformes, and Sphenisciformes as well as with marine mammals. Evaluating species infecting Phalacrocoracidae seems to illuminate the relationships among the cormorants (Hoberg 1987).

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# APPENDIX 1

**Partial list of metazoan parasites from the American white pelican.**

| Parasite | Site | Location[1] | ID[2] | Y/N[3] | Principal references |
|---|---|---|---|---|---|
| **Cestoda (Tapeworms)** | | | | | |
| Cyclustera ibisae | intestine | FL, TX | ID? | Y | Forrester and Spalding 2003, Dronen et al. 2003, Kinsella et al. 2004 |
| Diphyllobothrium cordiceps | intestine | WY, MT | | N | Leidy 1872, Woodbury 1932, Scott 1955, Post 1971 |
| Hymenolepis sp. | intestine | WY, WA | ID? | | Woodbury 1937, McNeil 1949 |
| Paradilepis longivaginosus | upper small intestine | WY | | N | Mayhew 1925, Woodbury 1937 |
| | | MB | | | McLaughlin 1974 |
| | | SD | | | Hugghins 1956 |
| | | CA-NV | | | Matthias 1963 |
| | | FL | | | Forrester and Spalding 2003, Kinsella et al. 2004 |
| | | MS, LA | | | Present study |
| Paradilepis caballeroi | intestine | FL | ID? | N | Forrester and Spalding 2003 |
| | | MS, LA | ID? | | Present study |
| Spirometra mansonoides spargana metacestode | subcutaneous | FL | | N | Forrester and Spalding 2003 |
| Tetrabothrius sp. | small intestine | TX | | Y | Dronen et al. 2003 |
| **Digenea (Flukes)** | | | | | |
| Ascocotyle gemina | intestine, ceca | FL | | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| Ascocotyle leighi | ceca, large intestine | FL | | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Austrobilharzia variglandis | intestine, mesentery vessels | MS | | N | Present study |
| Bolbophorus confusus (at least = B. damnificus in some references) | intestine | USA, Canada | ID? | Y | Hugghins 1956, Fox and Olson 1965, Dubois 1970, Dronen et al. 2003, Forrester and Spalding 2003 |
| Bolbophorus damnificus | intestine | MS, LA | | Y | Overstreet et al. 2002 |
| Bolbophorus sp. of Overstreet et al. 2002 | intestine | MS, LA OR | | N | Overstreet et al. 2002 Present study |
| Bursacetabulus pelecanus | intestine | FL MS, LA TX | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 Blend, Curran and Overstreet, personal observations Dronen et al. 2003 |
| Bursatintinnabulus macrobursus | intestine | TX, MS, LA FL | | Y | Blend, Curran and Overstreet, personal observations Forrester and Spalding 2003 |

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**APPENDIX 1 (continued)**

| Parasite | Site | Location[1] | ID[2] | Y/N[3] | Principal references |
|---|---|---|---|---|---|
| Digenea (Flukes) (continued) | | | | | |
| *Bursatintinnabulus bassanus* | ceca, intestine | TX | ID? | Y | Dronen et al. 2003 |
| | | FL | ID? | | Kinsella et al. 2004 |
| *Clinostomum attenuatum* | mouth cavity, trachea | FL | | | Kinsella et al. 2004 |
| *Clinostomum marginatum* | esophagus | LA | | N | Present study |
| | | FL, TX | | | Present study▼ |
| *Clinostomum complanatum* | mouth cavity, trachea | FL | ID? | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Dendritobilharzia pulverulenta* | blood vessels, heart | FL | | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Diplostomum spathaceum* | intestine | MB | ID? | N | McLaughlin 1974 |
| *Echinochasmus dietzevi* | intestine, stomach | FL | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 |
| | | MS, LA | | | Present study |
| *Gigantobilarzia huttoni* | veins of intestinal wall | FL | | Y | Leigh 1957 |
| *Galactosomum* sp. | small intestine | FL | | | Kinsella et al. 2004 |
| *Gigantobilarzia* sp. | blood vessels | FL | | | Forrester and Spalding 2003 |
| *Ignavia renalis* | intestine | London Zoo | | N | Yeh 1954 |
| *Mesostephanus appendiculatoides* | intestine, cloaca, ceca | FL | | Y | Forrester and Spalding 2003 |
| | ceca | MS, LA | | | Present study |
| *Mesostephanus microbursa* | small intestine | FL | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Mesostephanus* sp. | intestine, ceca | TX | ID? | | Dronen et al. 2003 |
| *Mesorchis denticulatus* | intestine, cloaca, ceca | MS, LA | | Y | Kinsella et al. 2004, present study |
| *Microparyphium facetum* | cloaca | FL | | | Kinsella et al. 2004 |
| *Phagicola longus* | small intestine | WA | | Y | McNeil 1949 |
| | cloaca, intestines | TX | | | Dronen et al. 2003 |
| | small intestine, ceca | FL | | | Kinsella et al. 2004 |
| | | MS, LA | | | Present study |
| *Phagicola nana* | small intestine | FL | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Pholeter anterouterus* | nodules in wall of small intestine | FL | | Y | Pearson and Courtney 1977, Kinsella et al. 2004 |
| *Posthodiplostomum mininum* | small intestine | FL | | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Prosthogonimus ovatus* | oviduct | OR | ID? | N | Present study |
| *Renicola thapari* | kidneys | FL | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 |
| | | MS | | | Present study |
| | | MN, CA | | | Present study▼ |

**APPENDIX 1 (continued)**

| Parasite | Site | Location[1] | ID[2] | Y/N[3] | Principal references |
|---|---|---|---|---|---|
| Digenea (Flukes) (continued) | | | | | |
| *Renicola* sp. | kidneys | OR | | N | Present study |
| | | FL | | | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Ribeiroia ondatrae* | esophagus, | WA | | N | McNeil 1949 |
| | proventriculus, | FL | | | Forrester and Spalding 2003, Kinsella et al. 2004 |
| | intestine | | | | |
| | | MS, LA | | | Present study |
| | | | | | |
| Acanthocephala (Spiny-headed Worms) | | | | | |
| *Polymorphus brevis*, immature | small intestine | FL | | N | Kinsella et al. 2004 |
| (not presently available) | intestine | MS | | ? | Present study |
| | | | | | |
| Nematoda (Roundworms) | | | | | |
| *Capillaria mergi* | intestines, ceca | FL | | Y? | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Capillaria* sp. | intestines, ceca | FL | | Y? | Forrester and Spalding 2003 |
| | | TX | | | Dronen et al. 2003 |
| *Contracaecum multipapillatum* | proventriculus, | MS, LA | | Y | Deardorff and Overstreet 1980, present study |
| | esophagus | | | | |
| | | FL | | | Kinsella et al. 2004 |
| *Contracaecum rudolphii* | proventriculus, | WY, SD | | Y | Woodbury 1937, Hugghins 1956 |
| (as *C. spiculigerum*) | esophagus | MS, LA | | | Deardorff and Overstreet 1980 |
| | | FL | | | Forrester and Spalding 2003, Kinsella *et al.* 2004 |
| *Contracaecum micropapillatum* | proventriculus | FL, MS, LA | | Y | Oglesby 1960, Deardorff and Overstreet 1980, |
| | | | | | Kinsella et al. 2004, present study |
| *Contracaecum microcephalum* | proventriculus, | TX | ID? | Y | McDaniel and Patterson 1966, |
| | esophagus | OR | | | Present study |
| *Contracaecum bancrofti* | proventriculus | TX | ID? | | McDaniel and Patterson 1966 |
| | | Mexico | ID? | | Yamaguti 1961 |
| *Contracaecum* spp. | esophagus, intestine | FL, TX | | Y? | Forrester and Spalding 2003, Dronen et al. 2003 |
| | stomach | | | | |
| *\*Cyathostoma phenisci* | trachea, lung | MS | | Y | Present study |

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| Parasite | Site | Location[1] | ID[2] | Y/N[3] | Principal references |
|---|---|---|---|---|---|
| *Eustrongylides ignotus*, juvenile | proventriculus | FL | | Y | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Gnathostoma spinigerum* | | Mexico | | | Camacho et al. 1998 |
| *Microtetrameres pelecani* | proventriculus | FL | | N | Forrester and Spalding 2003, Kinsella et al. 2004 |
| *Paracuaria adunca* | proventriculus | FL | | | Kinsella et al. 2004 |
| *Strongyloides* sp. | intestine | FL | | N | Forrester and Spalding 2003 |
| *Tetrameres* sp. | proventriculus | FL | | Y? | Forrester and Spalding 2003 |
| | | | | | |
| Hirudinea (Leeches) | | | | | |
| *Theromyzon* sp. | feathers from breast to head | OR | | N | Present study |
| | | | | | |
| Phthiraptera (Chewing Lice) | | | | | |
| *Colpocephalum unciferum* | feathers | SD, Canada, | | N | Kellogg 1896, Hugghins 1956, |
| | | KS | | | Price1967 |
| | | BC, MB, SK | | | Price and Beer 1967 |
| | | FL | | | Forrester et al. 1995 |
| | | MS, LA, OR | | | Present study |
| *Pectinopygus tordoffi* | feathers | KS, UT, SD, | | N | Elbel and Emerson 1956 |
| | | W DC, | | | |
| | | MS, TX, CA, OR | | | |
| | | Mexico | | | Carriker 1956 |
| | | FL | | | Forrester et al. 1995 |
| | | MS, LA, OR | | | Present study |
| *Piagetiella peralis* | gular pouches and body | FL | | N | Leidy 1878, Forrester et al. 1995 |
| | | WAMc | | | Neil 1949 |
| | | SD | | | Hugghins 1956 |
| | | SK | | | Wobeser et al. 1974 |
| | | MS, LA, OR | | | Present study |
| | | | | | |
| Siphonaptera (Fleas) | | | | | |
| *Ceratophyllus niger* | body and nest | BC | | Y | Holland 1985 |
| | | | | | |
| Diptera (Flies) | | | | | |

**APPENDIX 1 (continued)**

| Parasite | Site | Location[1] | ID[2] | Y/N[3] | Principal references |
|---|---|---|---|---|---|
| Hippoboscidae (Louse Flies) | | | | | |
| *Icosta albipennis* | body | USA | | | Maa 1969 (listed as probably accidental, pelican species not identified) |
| *Olfersia sorida* | body | USA? | ? | | Maa 1969 |
| Calliphoridae (Blowflies) | | | | | |
| *Bufolucilia silvarum* | body | ND | | | Johnson 1976 |
| *Lucilia illustris* | body | ND | | | Johnson 1976 |
| *Phaenicia sericata* | body | ND | | | Johnson 1976 |
| *Phormia regina* | body | ND | | | Johnson 1976 |
| Muscidae (House Flies, Stable Flies) | | | | | |
| *Graphomya americana* | body | ND | | | Johnson 1976 |
| *Musca autumnalis* | body | ND | | | Johnson 1976 |
| *Stomoxys calcitrans* | body | ND | | | Johnson 1976 |
| (Class) Arachnida | | | | | |
| Argasidae (Soft Ticks) | | | | | |
| *Ornithodoros capensis* | pouch, body, and nest | TX | | Y | King et al. in Duffy 1983 |
| Hypoderatidae (Nest Mites) | | | | | |
| *Phalacrodectes* sp.? | subcutaneous, trachea and neck | MS | ID? | Y | Present study (specimens not available) |
| Analgoidea (Feather Mites) | | | | | |
| *Metingrassia pelecani* | feathers | Canada | | | Mironov and Galloway, 2002 |
| *\*Plicatalloptes pelecani* (as *Alloptes* sp.?) | feathers | Canada, MS / FL | | N | Mironov, personal communication / Forrester and Spalding 2003 |
| *\*Scutomegninia gaudi* (as *Scutomegninia* sp.?) | feathers | Canada, MS / FL | | N | Mironov, personal communication / Forrester and Spalding 2003 |

1 = Abbreviations refer to states in the USA and provinces in Canada.

2 ID? = Questionable identification, see text for explanation.

3 Y/N = Has (Y) or has not (N) also been reported from the brown pelican.

\* = New host record.

▸ = Based on specimens loaned to us by the National Wildlife Health Research, Madison, Wisconsin.

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## Ecology of the Rock Shrimp *Sicyonia dorsalis* Kingsley, 1878 (Crustacea: Sicyoniidae) in a Subtropical Region of Brazil

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# ECOLOGY OF THE ROCK SHRIMP *SICYONIA DORSALIS* KINGSLEY, 1878 (CRUSTACEA: SICYONIIDAE) IN A SUBTROPICAL REGION OF BRAZIL

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*ABSTRACT* The present study analyzes the abundance and distribution of the rock shrimp *Sicyonia dorsalis*, in relation to water temperature, salinity, depth, organic matter content, and sediment texture in Mar Virado (MV), Ubatuba (UBA) and Ubatumirim (UBM), 3 distinct bays along the northern coast of São Paulo State (23°S, 45°W), Brazil. Six transects were taken in each bay, 4 being parallel to the coastline and 2 next to the rocky shores. Monthly samples were taken over a 2-year period (1998 and 1999) with a shrimp fishing boat equipped with double-rig nets. A total of 2,498 specimens was obtained with 804 from MV, 922 from UBA, and 772 from UBM. The spatial distribution of *S. dorsalis* did not differ among bays. Higher abundance values were recorded in areas where silt+clay comprised more than 60% of the sediment. Abundance also followed a seasonal trend, being highest during spring when intrusions of the cold South Atlantic Coastal Waters are most common, promoting the migration of this shrimp species to more sheltered areas. In short, sediment type and water temperature appear to be the most important environmental variables analyzed which affect the spatial and seasonal distribution of *S. dorsalis*.

## INTRODUCTION

Sicyoniid shrimps are represented on the Brazilian coast by 6 species (D'Incao 1995): *Sicyonia dorsalis* Kingsley, 1878, *Sicyonia typica* (Boeck, 1864), *Sicyonia laevigata* Stimpson, 1871, *Sicyonia parri* (Burkenroad, 1934), *Sicyonia burkenroadi* Cobb, 1971 and *Sicyonia olgae* Pérez Farfante, 1980. Among these, *S. dorsalis, S. typica, S. laevigata,* and *S. parri* occur in the southeastern subtropical region of Brazil (Costa et al. 2000). *Sicyonia dorsalis* is distributed from Cape Hatteras, North Carolina (USA) including GOM to Florianópolis, Santa Catarina (Brazil) (Pérez Farfante and Kensley 1997). This species has been found from mouth of bays to 60 m deep, rarely to 420 m (Williams 1984) and, according to D'Incao (1995), the highest abundance is at 80 m. This non-commercial species along the southeastern Brazilian coast, due to its small size, constitutes the highest percentage (92%) of captured sicyoniid (Costa 2002). With respect to other Penaeoidea found in the present studied region, *S. dorsalis* is the 7th (1%) most abundant species, about 90% of the shrimps are the seabob, *Xiphopenaeus kroyeri* followed by *Farfantepenaeus brasilensis, F. paulensis, Litopenaeus schmitti, Artemesia longinaris, Rimapenaeus constrictus,* and *Pleoticus muelleri* (Costa 2002).

Most accounts of *S. dorsalis* biology are based on populations studied in the northern hemisphere. Emphasis has been given to reproductive aspects (Bauer 1992, 1996a,b) and composition, abundance and diversity patterns within the benthic community (Wenner and Boesch 1979, Wenner and Read 1982, Sánchez and Soto 1987). These latter authors also verified the influence of some abiotic factors on the distribution of penaeoidean shrimps, including *S. dorsalis*.

Biological studies on *S. dorsalis* along the Brazilian coast are limited to a few biogeographical records and some aspects of their ecology. D'Incao (1995) presented information on taxonomy and geographic and bathymetric distributions based on specimens from scientific collections. Costa et al. (2000, 2003) also verified the presence of this species during faunistic surveys of shrimps in the Ubatuba region, State of São Paulo, Brazil. Fransozo et al. (2002) observed seasonal abundance patterns of some penaeoidean shrimps. However, the distribution in the Brazilian coast of *S. dorsalis* relative to ecological factors has not been studied to date. The aim of this study was to determine the spatial and seasonal distribution of *S. dorsalis* in the bays of Mar Virado (MV), Ubatuba (UBA), and Ubatumirim(UBM) in relation salinity, temperature, depth, sediment texture, and organic matter content in the Ubatuba region.

## MATERIALS AND METHODS

Shrimps were collected monthly during day hours from January 1998 to December 1999 at MV, UBA, and UBM bays, located in the Ubatuba region, São Paulo State. Collections were made during daylight as Negreiros-Fransozo *et al.* (1999) verified that the abundance of *S. dorsalis* was not correlated with light intensity. Each bay
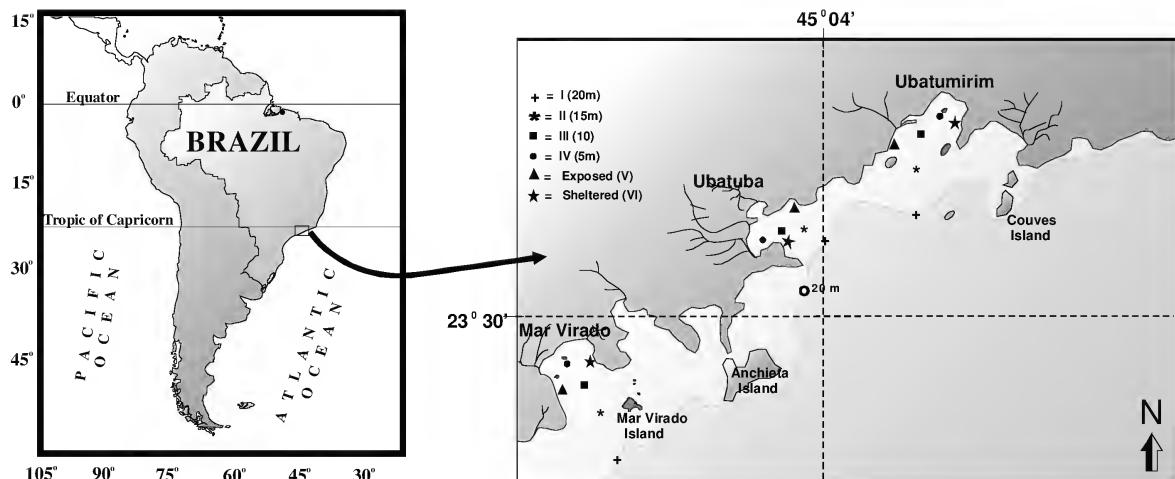
**Figure 1. Study region indicating collection sites.**

was divided into 6 transects (2 km each) and trawled over a 30-min period (Figure 1). Four transects were located at mean depths of 5 (IV), 10 (III), 15 (II) and 20 m (I), and the other 2 adjacent to rocky shores on exposed (V) and sheltered (VI) shore. A shrimp fishing boat equipped with 2 double rig nets (mesh size 20 mm and 15 mm in the cod end) was used for trawling. During the study period 422 trawls were conducted (144 in each bay).

Salinity (psu) and water temperature (°C) were measured from bottom-water samples obtained each month for each transect using a Nansen bottle. An ecobathymeter coupled with a Global Position System was used to record depth at sampling sites. Sediments samples were collected in each season with a 0.06 m² Van Veen grab. Grain size categories followed the American standard, for which sediments were sieved at 2.0 mm (gravel), 1.0 mm (very coarse sand), 0.5 mm (coarse sand), 0.25 mm (intermediate sand), 0.125 mm (fine sand), and 0.0625 mm (very fine sand) smaller particles were classified as silt-clay. Grain size fractions were expressed in the phi ($\varnothing$) scale, thus accounting for the central tendency of sediment samples, e.g., -1= phi < 0 (gravel); 0 = phi < 1 (coarse sand); 1 = phi < 2 (intermediate sand); 2 = phi < 3 (fine sand); 3 = phi < 4 (very fine sand) and phi = 4 (silt + clay). Cumulative particle size curves were plotted using the phi-scale, and phi values corresponding to 16th, 50th, and 84th percentiles were read from the curves to determine the mean diameter of the sediment. This was calculated according to the formula: ($\varnothing$16 + $\varnothing$50 + $\varnothing$84)/3, after that, the phi was calculated from the formula $\varnothing = -\log_2 d$, where $d$ = grain diameter (mm). All procedures employed for sediment analysis followed Hakanson and Jansson (1983) and Tucker (1988).

The kind of the ecological distribution of *S. dorsalis* was analyzed using the Kolmogorov-Smirnov test ($P <$ 0.01). The abundance of shrimps were compared among years, bays, transects and seasons of the year using analysis of variance (ANOVA, $P < 0.05$). The influence of environmental factors on *S. dorsalis* abundance were evaluated by multiple linear regression and also compared through ANOVA ($P < 0.05$). Data were $\log_{10}$-transformed prior to the analysis to improve their normality (Zar 1999).

## RESULTS

The mean depth of each transect in the bays sampled was I (22.2 ± 0.6m), II (16.5 ± 1.1m), III (11.6 ± 1.1m), IV (5.9 ± 0.4m), V (9.2 ± 1.5m) e VI (6.8 ± 2.3m). In general, mean grain size (phi) of sediments varied from intermediate sand to silt+clay. The amount of mud in the sediments decreased northward within the sampled areas from MV to UBM (Table 1).

In MV bay, the silt + clay fraction (phi > 4) dominated all transects, comprising more than 75% of the samples (Table 1). The phi values decreased in each transect of other sampled bays. In UBA, and mainly in UBM bay, a predominance of fine and very fine sand, associated with silt+clay was observed (Table 1), except at transect I in UBM (phi = 1.5). The organic matter content in the substratum was lowest in the offshore region (transect I and II) of the 3 bays, whereas I was the highest in the other transects (Figure 2).

There was a clear water temperature difference among transects during spring and summer. Water temperature at transects I through III was lower than at transects IV through VI (Figure 3). During other seasons, the mean

**TABLE 1**

**The mean diameter of the sediment (phi), quantity of mud (% silt+clay) and mean number of *Sicyonia dorsalis* by trawl (*n*) at each transect at each bay sampled during 1998 and 1999.**

| | Bays | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Mar Virado | | | Ubatuba | | | Ubatumirim | | |
| Transects | phi (Ø) | % mud | *n* | phi (Ø) | % mud | *n* | phi (Ø) | % mud | *n* |
| I | 4.3 | 46.8 | 4.2 | 3.2 | 16.0 | 0.3 | 1.5 | 2.6 | 0.1 |
| II | 5.7 | 75.3 | 3.5 | 3.99 | 21.2 | 0.8 | 3.8 | 23.9 | 0.6 |
| III | 6.2 | 88.3 | 6.7 | 5.3 | 61.9 | 8.8 | 4.4 | 35.7 | 17.3 |
| IV | 5.9 | 81.2 | 1.5 | 5.7 | 76.3 | 15.7 | 4.9 | 49.6 | 1.4 |
| V | 5.8 | 79.7 | 6.0 | 4.8 | 47.3 | 3.9 | 4.0 | 22.2 | 7.6 |
| VI | 5.4 | 64.4 | 11.7 | 3.6 | 36.8 | 9.0 | 4.4 | 33.4 | 5.2 |
| Total | | | 5.6 | | | 6.4 | | | 5.4 |

water temperature values were homogeneous. Variation in the mean bottom salinity within each bay is shown in Figure 4. Differences in salinities between bays is substantial with the lowest mean values recorded in MV. In general, higher salinity ranges were found at transect I, whereas lower salinities were found at transects IV and VI (Figure 4).

A total of 2,498 shrimps was obtained, 1,385 during the first year, and 1,113 during the second year. The analysis of shrimp distribution reveals they are contagiously distributed in the studied area ($P < 0.01$). For the pooled sample, the absolute abundance was highest in UBA (922), followed by MV (804) and UBM (772). The comparison of shrimp abundance among bays, years, transects and seasons is shown in Table 2. No significant difference in abundance was found between years or among bays ($P < 0.05$; Table 2).

*Sicyonia dorsalis* was more abundant along transects VI in MV bay and IV in UBA Bay than along other transects. In UBM, catches were highest along transect III. In general, the lowest number of specimens were collected along transects I and II, at each bay (Figure 5). Significant differences were obtained among transcripts ($P < 0.05$, Table 2).

The highest shrimp abundance occurred during spring (October to December) 1998 and early summer (January and February) and the early summer and late spring (November and December) 1999. These periods had significantly higher shrimp than other seasons ($P < 0.05$, Table 3). Conversely, lowest abundances occurred during fall and winter.

There was a good fit between *S. dorsalis* and 5 environmental variables, and this relationship is explained by Abundance $= 11.78 - 1.67$ (bottom temperature) $- 4.63$ (bottom salinity) $- 0.03$ (% organic matter) $- 0.13$ (depth) $+ 0.97$ (phi) ($r = 0.30$, $P = 1.0001$ E$^{-7}$, $n = 432$). Water temperature and salinity were negatively associated and mud content (phi) was positively associated with the number of individuals. However, no correlation was observed between organic matter content and depth in the distribution of this species ($P > 0.05$, Table 4). The analysis indicated that more individuals were collected in conditions of higher percentage of silt and clay, bottom temperature between 19 and 22 °C and salinity between 30 and 34 psu) (Table 1, Figure 6). Also, there greater numbers of *S. dorsalis* in depths < 15m in spring and summer (Figure 7), following a decrease in bottom water temperature (Figure 3). In fall and winter, there was more homogeneity in the spatial distribution of *S. dorsalis*, however the abundance was lower than other seasons.
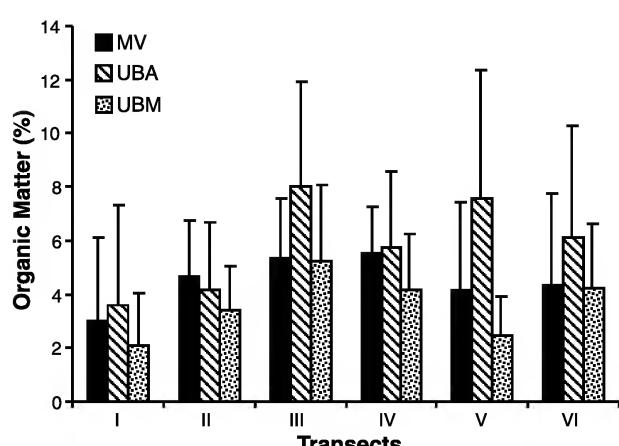


Figure 2. Mean values (±sd) of organic matter content in sediments (%) at each sampled transect in bays studied. MV = Mar Virado, UBA = Ubatuba, UBM = Ubatumirim.
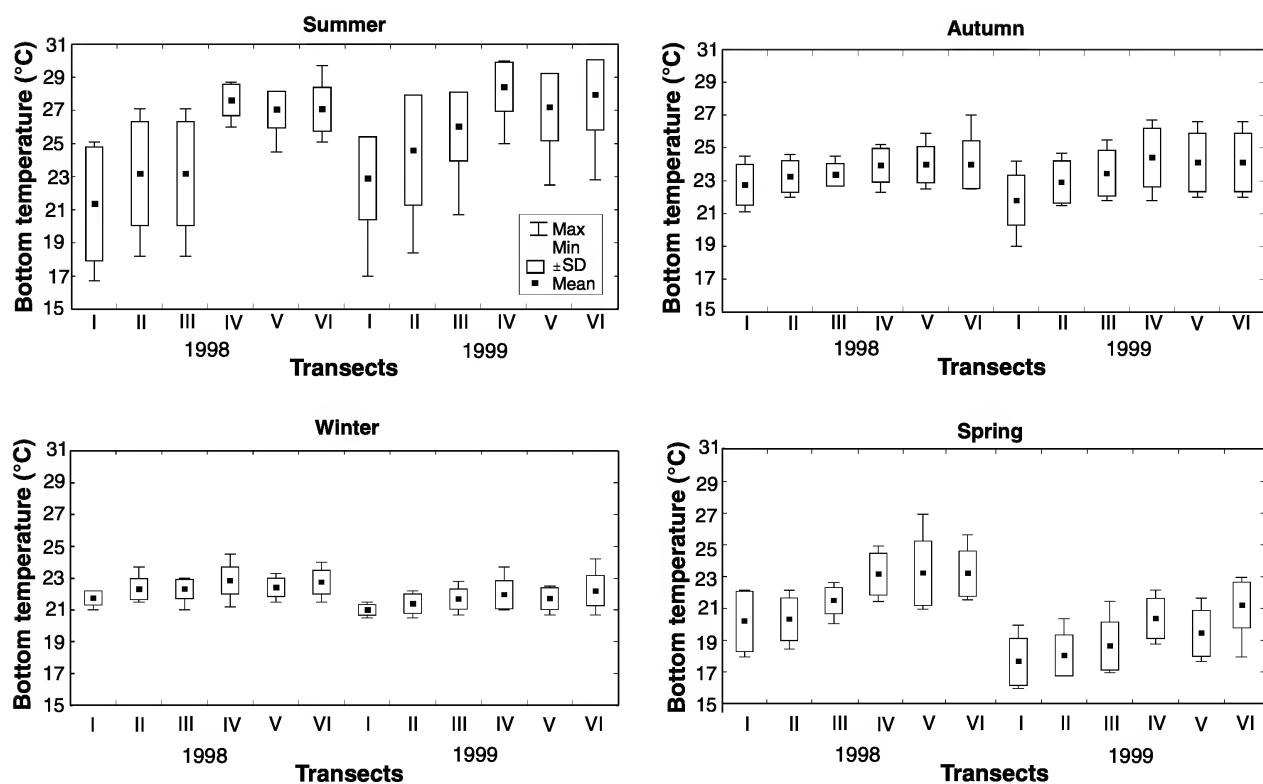
Figure 3. Boxplots showing mean, standard deviation, maximum and minimum temperature values (°C) for each transect and season in 1998 and 1999.
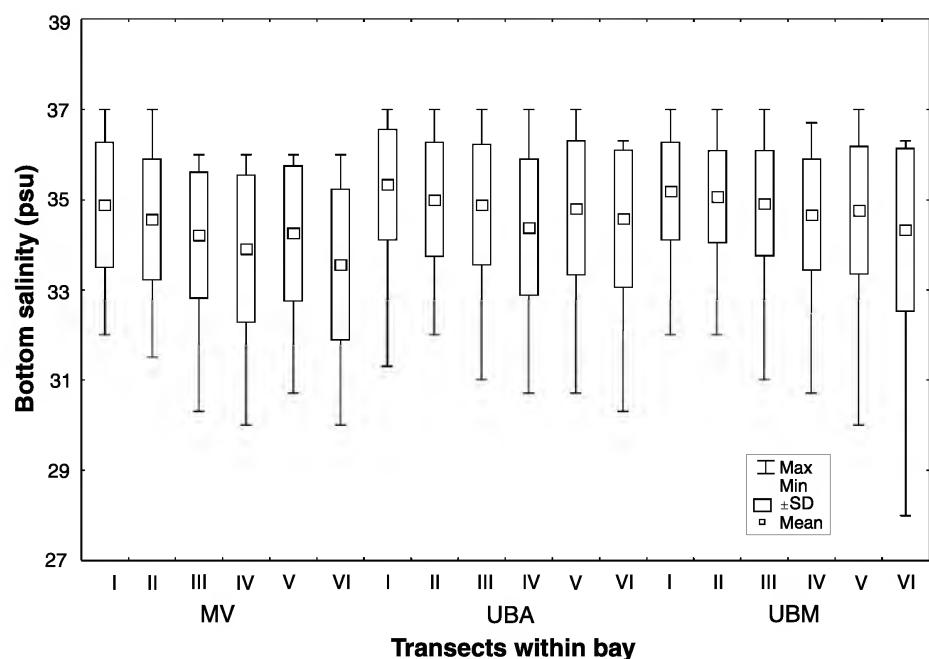


Figure 4. Boxplots showing mean, standard deviation, maximum and minimum salinity values (psu) for each transect within each bay in 1998 and 1999. MV = Mar Virado, UBA = Ubatuba, UBM = Ubatumirim.

## TABLE 2

**Results of the analysis of variance of the mean catch (data $\log_{10}$-transformed) of *Sicyonia dorsalis* by year, bay, transect or season.**

| Source | df | MS | F | p |
|---|---|---|---|---|
| Bay | 2 | 2.21 | 2.47 | 0.0855 |
| Transect (bay) | 15 | 6.42 | 7.20 | 0.0001 |
| Year | 1 | 1.24 | 1.38 | 0.2403 |
| Season | 3 | 37.71 | 42.27 | 0.0001 |
| Season x Year | 3 | 9.23 | 10.34 | 0.0001 |

## DISCUSSION

Castro-Filho et al. (1987) showed that the study region is strongly influenced by 2 types of water currents: coastal waters (CW) and tropical waters (TW). These currents occur during fall and winter, causing an increase in water temperature and salinity to over 21 °C and 35 psu, respectively. Also, another current occurs throughout late spring and summer, the South Atlantic Central Water (SACW), causing decreases in water temperature (< 20 °C) and bottom salinity (< 35). The incursion of the TW into the uppermost water layers and the dislocation of the CW towards the ocean during the fall and winter cause vertical mixing and thus eliminate the existing seasonal thermocline, causing the SACW to recede towards the offshore region.

The intrusion of SACW was detected in this study during spring and summer at 10 and 15 m isobaths. Our results indicate that fluctuations in the seasonal and bathymetric distribution of *S. dorsalis* were influenced by variation in water temperature caused by these currents. When intruding into the bays, SACW causes a decrease in water temperature and confinement of the shrimp in shallower areas (< 15m). Similar results were also reported for the shrimp *Xiphopenaeus kroyeri* (Heller, 1862) by Nakagaki and Negreiros-Fransozo (1998), *Rimapenaeus constrictus* (Stimpson, 1874) by Costa and Fransozo (2004), and the "argentinean" shrimp *Pleoticus muelleri* Bate, 1888 by Costa et al. (2004), all in Ubatuba bay. In contrast, during late summer and autumn, when bottom-water temperature increased, a few specimens were captured in shallower areas. It may be inferred that the elevation of water temperature during these periods caused the migration of shrimp to the outer areas of the bays

In spite of the association found between the abundance of *S. dorsalis* and low salinity conditions, there is no evidence of a direct influence of salinity in the distribution of this species. Past biological studies on *S. dorsalis* were restricted to bathymetric distribution, and occurrences on sediment type (Williams 1984, Sanchez and Soto 1985, D'Incao 1995). Only Gunter (1950) and Fransozo et al. (2002) have focused on the influence of salinity and stated that *S. dorsalis* were captured in areas where salinity was > 33.5 psu. However, Gunter (1950) found only 10 individuals 8.05 km offshore in the Gulf of Mexico in Texas, and Fransozo et al. (2002) found 35 individuals in Fortaleza bay, Ubatuba, São Paulo. According to our results and the bathymetric distribution mentioned for this species, we suggest that it prefers areas with values above 30 psu. Pérez Farfante (1985) also pointed out that other congeneric species such as *S. brevirostris* Stimpson 1871 and *S. ingentis* (Burkenroad, 1938) occur in waters of high salinity (33 to 35 psu) and that these shrimps do not depend upon estuarine waters for their life cycle.
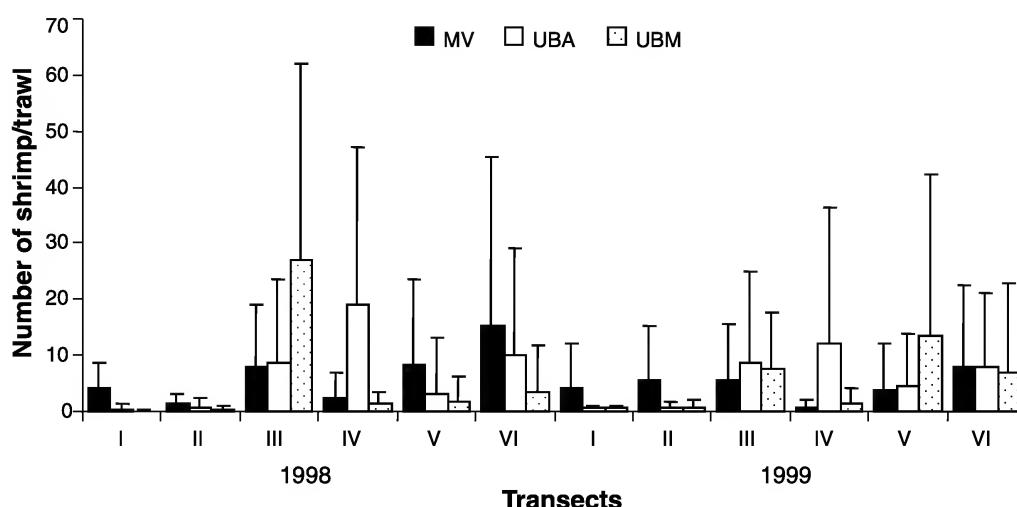


Figure 5. Mean number of shrimp by bay and transect during 1998 and 1999. MV = Mar Virado, UBA = Ubatuba, UBM = Ubatumirim.

## TABLE 3

Monthly catch of *Sicyonia dorsalis* with each bay in 1998, 1999, and total catch for each season. Results of the ANOVA are shown for each season. Abundance followed by the same letter in the column (Season) do not differ statistically (*P* > 0.05). MV = Mar Virado; UBA = Ubatuba; UBM = Ubatumirim.

| Month | Bays | | | | | | Total | | Season |
| | MV | | UBA | | UBM | | | | |
| Year | 98 | 99 | 98 | 99 | 98 | 99 | 98 | 99 | Total |
|---|---|---|---|---|---|---|---|---|---|
| January | 1 | 112 | 20 | 61 | 8 | 35 | 29 | 208 | Summer/98 = 58 a |
| February | 9 | 64 | 10 | 80 | 2 | 28 | 21 | 172 | Summer/99 = 394 b |
| March | 3 | 8 | 5 | 3 | 0 | 3 | 8 | 14 | |
| April | 1 | 0 | 2 | 2 | 2 | 32 | 5 | 34 | Fall/98 = 38 a |
| May | 2 | 4 | 1 | 4 | 5 | 2 | 8 | 10 | Fall/99 = 45 a |
| June | 2 | 0 | 11 | 0 | 12 | 1 | 25 | 1 | |
| July | 8 | 4 | 21 | 6 | 25 | 3 | 54 | 13 | Winter/98 = 251 b |
| August | 44 | 4 | 34 | 26 | 23 | 1 | 101 | 31 | Winter/99 = 66 a |
| September | 32 | 4 | 34 | 12 | 30 | 6 | 96 | 22 | |
| October | 40 | 19 | 44 | 31 | 21 | 7 | 105 | 57 | Spring/98 = 1038 c |
| November | 85 | 50 | 31 | 26 | 81 | 98 | 197 | 174 | Spring/99 = 608 bc |
| December | 240 | 68 | 296 | 162 | 200 | 147 | 736 | 377 | |
| Total | 467 | 337 | 509 | 413 | 409 | 363 | 1385 | 1113 | 2498 |

The abundance of *S. dorsalis* in the bays does not differ statistically, although it was higher in UBA bay, followed by MV bay. This probably results from the higher content of silt and clay in those areas. The more sorted sediments in the other sites, as for transect III in UBM and transect VI in UBA, are apparently preferred by this species, and appear to favor establishment of populations. Similar results were obtained by Sánchez and Soto (1987) for a population of *S. dorsalis* in the Gulf of Mexico and Pérez Farfante (1985) for the geminate species, *S. disdorsalis* (Burkenroad, 1934), in the eastern Pacific where shrimps were found associated with muddy sediments.

In transects where the number of shrimps was highest, besides the prevalence of silt + clay, we observed that these sites are located in a more sheltered area of each bay. Because of the particular hydrodynamics acting on these areas, water currents are weak at transects VI in MV bay,

at transects III and V in BM and transects III, VI, and mainly IV, in UBA bay. This favors the deposition of fine sediments, and consequently allowing settlement of *S. dorsalis*.

## TABLE 4

Results of a multiple linear regression among environmental factors and the number of *Sicyonia dorsalis*.

| Environmental factors | t | *P* |
|---|---|---|
| bottom temperature (°C) | −2.951 | 0.0033 |
| bottom salinity (psu) | −2.918 | 0.0037 |
| depth (m) | −0.808 | 0.4200 |
| organic matter | −0.338 | 0.7360 |
| phi (f) | 3.585 | 0.0004 |
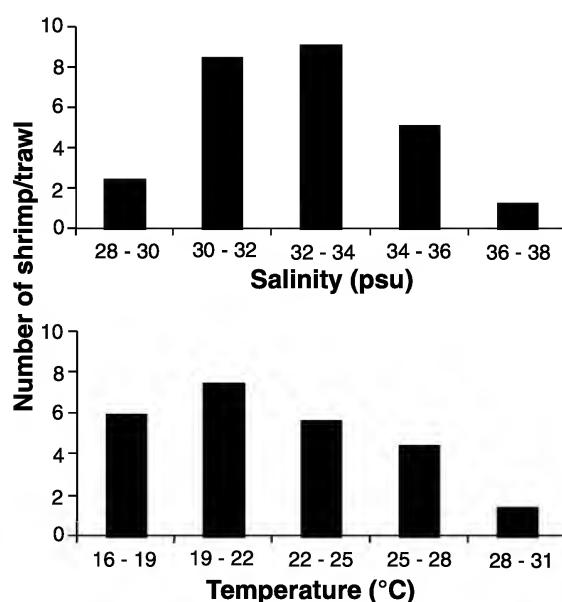


Figure 6. Plot of the mean number of shrimp in each salinity and water temperature class per trawl.
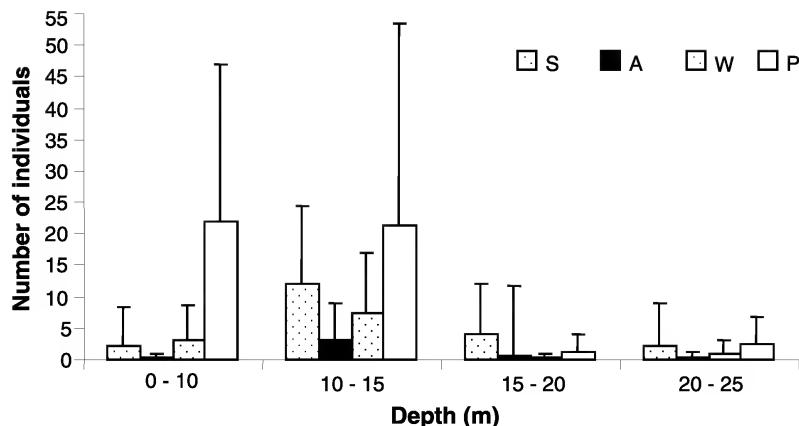
**Figure 7. Distribution of the mean (±sd) number of shrimp by depth class per trawl by season (S = summer, A = autumn, W = winter, P = spring).**

The spatial distribution of many penaeidean shrimps is mainly influenced by texture and organic content of the substratum (Dall et al. 1990). However, the organic matter content of sediments does not seem to affect the distribution of *S. dorsalis* for this area. The data obtained in the present study has confirmed the influence of texture of the sediment, therefore the distinct features of the sediment in each bay contributes in a significant way to the occurrence of the shrimps along the studied region. The distribution of the penaeids species *Metapenaeus macleayi* (Hasweell, 1879), *Penaeus monodon* Fabricius, 1798, *Penaeus esculentus* Haswell, 1879, *P. semisulcatus* De Haan, 1884 and *R. constrictus* are more influenced by grain size than by the availability of food (Ruello 1973, Brandford 1981, Somers 1987, Costa and Fransozo 2004). Although water temperature and sediment type offer a most convincing explanation for distributional patterns of *S. dorsalis*, it is important to realize that other factors such as diurnal and nocturnal variation, competition and predation may also influence its distribution.

## ACKNOWLEDGMENTS

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Distribution of Myodocopid Ostracods in Tampa Bay, Florida, and Association with Abiotic Variables

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# DISTRIBUTION OF MYODOCOPID OSTRACODS IN TAMPA BAY, FLORIDA, AND ASSOCIATION WITH ABIOTIC VARIABLES

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*ABSTRACT* Myodocopid ostracods were identified from > 600 benthic samples collected from Tampa Bay, Florida, during 1995 to 2001, as part of an annual synoptic survey of the benthos. At least 24 taxa were present. *Parasterope pollex* was the most abundant (76%) and most frequently collected (48%) species; *Rutiderma darbyi* (28%) and *Eusarsiella disparalis* (16%) were the next most frequently collected species. Logistic regression and "center of abundance" calculations were used to identify habitat "preferences" for the most frequently occurring species. With the exception of *P. pollex*, these were more likely to occur in coarser sediments, in more saline waters, and at greater depths than the mean for Tampa Bay. *Parasterope pollex* occurred over the widest ranges of salinity and sediment types, although it preferred medium to fine sand-sized sediments; *P. pollex* was also the species most tolerant of low dissolved oxygen concentrations.

## INTRODUCTION

Myodocopid ostracods are common inhabitants of estuarine and marine sediments, although species-specific quantitative ecological information is often lacking. The different families of myodocopids appear to serve different roles in energy transfer. Filter feeding is believed to be typical of the Cylindroleberidae, detritivory of the Philomedidae; the Cypridinidae include scavengers, and both the Sarsiellidae and Rutidermatidae appear to be predators (Cannon 1933, Kornicker 1975, Vannier et al. 1998). Many species are capable of migrating into the water column (Schram 1986, Alldredge and King 1985), where they may serve as prey for zooplankton (Vannier et al. 1998) and fish (Hobson and Chess 1976).

At least 34 species of myodocopid ostracods have been identified as occurring in shallow, near-shore waters of peninsular Florida, including the Gulf of Mexico (GOM) (Kornicker 1977, 1983, 1984a, 1984b, 1986a, 1986b, Kornicker and Iliffe 1989, Horsley 1990, Grabe et al. 1995, Kornicker and Grabe 2000). With the exception of work done in southwest Florida (Grabe et al. 1995), these papers primarily address the taxonomy of myodocopids. Ecological information on myodocopids is often ancillary to the species descriptions.

This study examines the spatial distribution and taxonomic composition of myodocopid ostracods in Tampa Bay, Florida, one of the largest estuaries in Florida (> 1,000 km$^2$ ; Clark and Macauley 1989). Habitat preferences are quantified for the more frequently occurring species. Representative specimens from these collections are deposited in the US National Museum.

## MATERIALS AND METHODS

The study employed a stratified (by 7 bay segments) probabilistic design (Larsen et al. 1994; Coastal Environmental, Inc. 1994). Hexagonal grids were randomly superimposed over the Tampa Bay estuarine system. Within each hexagon, the sampling location was randomly determined, with a known probability of inclusion. Bay segments included Boca Ciega Bay, Hillsborough Bay, Lower Tampa Bay, the Manatee River, Middle Tampa Bay, Old Tampa Bay, and Terra Ceia Bay (Figure 1). Although the program commenced in 1993 and continues to the present, ostracods were analyzed only from 610 samples collected during 1995 to 2001 (Figure 1). All sampling occurred during late July–early October.

Benthic infauna, hydrographic profiles, and sediments were collected using the standard EMAP techniques adopted by USEPA for the Louisianan Province (Holland 1990). At each station, the water column profile for temperature (°C), dissolved oxygen (DO; mg/l), and salinity (psu) was measured with a Hydrolab Surveyor 3.

Sediment samples were collected with a 0.04 m$^2$ Young sampler. A core was removed from each sample and stored, on ice, for subsequent characterization of the sediment. Benthic samples were stored, on ice, after adding a solution of magnesium sulfate to relax the organisms. Samples were later sieved (0.5 mm mesh) and then fixed in a 10% solution of borax-buffered formalin and Rose Bengal.

Ostracods were not a taxon of interest to the bay-wide benthic monitoring program. However, the myodocopid ostracods were removed from most of the samples collected during 1995 to 2001 and identified to the lowest practical taxonomic level.
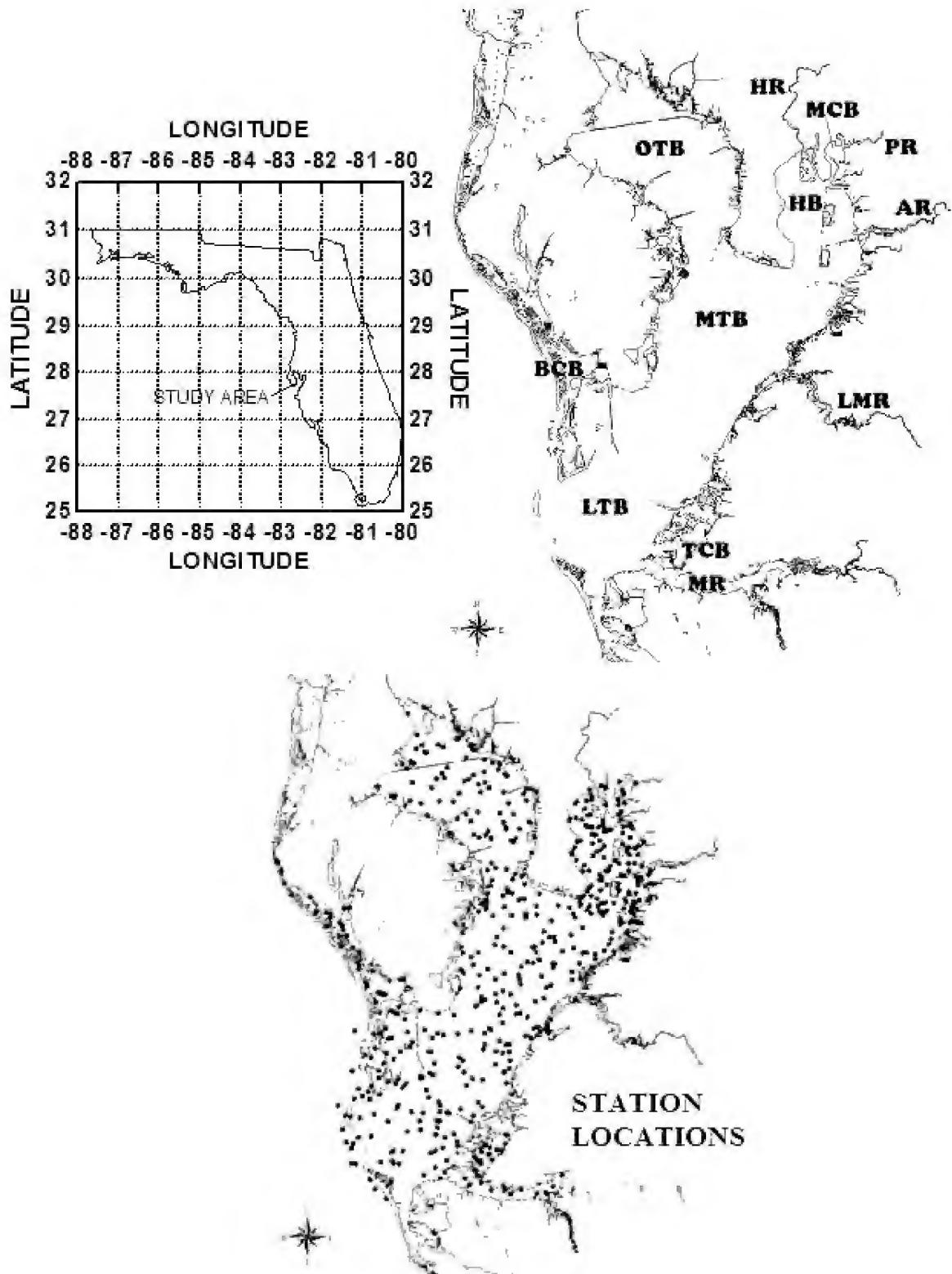
Figure 1. Location of sampling stations for myodocopid ostracods in Tampa Bay, Florida, 1995-2001. Bay segments are BCB (Boca Ciega Bay), HB (Hillsborough Bay), LTB (Lower Tampa Bay), MR (Manatee River), MTB (Middle Tampa Bay), OTB (Old Tampa Bay), and TCB (Terra Ceia Bay). Subareas are AR (Alafia River), HR (Hillsborough River), LMR (Little Manatee River), MCB (McKay Bay), and PR (Palm River).

**TABLE 1**

**Mean, median, and range of selected abiotic variables by relative depth, from Tampa Bay, Florida 1995–2001.**

| Variable | Mean | Median | Range |
|---|---|---|---|
| Depth (m) | 2.8 | 2.5 | 0.1–13.2 |
| Silt+Clay (%) | 8.4 | 4.4 | 0.1–91.8 |
| Temperature-Surface (°C) | 28.9 | 29.0 | 21.6–39.2 |
| Temperature-Bottom (°C) | 28.7 | 28.7 | 21.6–39.2 |
| Salinity-Surface (psu) | 25.2 | 26.1 | 2.4–35.9 |
| Salinity-Bottom (psu) | 26.1 | 26.9 | 4.3–36.0 |
| Dissolved Oxygen-Surface (mg/l) | 6.2 | 6.1 | 1.1–13.2 |
| Dissolved Oxygen-Bottom (mg/l) | 5.2 | 5.4 | 0.2–14.0 |

Sediments were analyzed to determine the percentage of silt+clay (%SC) particles < 63 μm diameter. An aliquot of sediment was wet sieved through a 63 μm mesh sieve and dried to a constant weight (Strobel et al. 1995).

Data collected by Long et al. (1994) from Tampa Bay were used to estimate a relationship between %SC and mean grain ($\phi$) using TableCurve 2D (SYSTAT 2002):

$$\%SC = 1/(0.0097 + 1.575 \quad e^{\phi}); \text{ (adjusted } r^2 = 0.947)$$

$\phi$ was then estimated for each %SC value from the 1993-2001 samples. Sediments were then categorized (e.g., medium sand, mud) according to the Wentworth scale breakpoints for $\phi$ (cf. Percival and Lindsay 1997).

The percent similarity of species associations were examined using the Sorensen coefficient for presence-absence (Clarke and Warwick 2001). Logistic regression was used to characterize habitat preferences for the 10 most frequently occurring species (cf. Huisman et al. 1993, Peeters and Gardiniers 1998, Ysebaert et al. 2002). Forward stepwise multiple logistic regression (SPSS, Inc. 2000) was used to identify abiotic variables best able to predict the occurrence of the 10 species. $\text{Log}_{10}$ $(n + 1)$ transformed abiotic variables used in this analysis include depth (m), salinity, temperature, DO, and %SC (arc sine (ASN)). TableCurve 2D (SYSTAT 2002) was used to develop univariate Gaussian logistic regression equations so that the "optimum" value and the "tolerance" (preferred range) could be calculated (Peeters and Gardiniers 1998). McFaddens's Rho$^2$ was used as a measure of goodness-of-fit (McFadden 1974, Hensher and Johnson, 1981). Values are similar to, but generally lower than, the coefficient of determination. Hensher and Johnson (1981) suggest that values between 0.2 and 0.4 represent a good fit.

Results of these analyses should be treated cautiously as the sample sizes are small relative to those used by Peeters and Gardiniers (1998) and Ysebaert et al. (2002).

Center of abundance calculations were also made: $\Sigma$(species abundance abiotic variable)/$\Sigma$ species abundance.

## RESULTS AND DISCUSSION

### Study area

Sample depths during the study ranged from 0.1 to 13.2 m, although the median depth was 2.5 m (Table 1). Near-bottom salinities in Tampa Bay during the summer-fall period are typically in the polyhaline (18–30 psu) range (Table 1). Sediment types in Tampa Bay included coarse sands (including shell hash; < 1.7 %SC), medium sands ( 1.7 < 4.51 %SC), fine sands ( 4.51 < 11.35 %SC), very-fine sands (> 11.35 < 25.95 %SC), and mud-sized sediments (> 25.95 %SC). Tampa Bay sediments are predominantly medium to fine sand-sized sediment, although mud-sized sediments are located in tributaries and portions of Hillsborough Bay (Figure 2). Near-bottom DO concentrations in the bay are generally above 4 mg/l, although mesohaline and polyhaline very fine sand and mud habitats were often hypoxic.

### Overview of the mydocopid assemblage

At least 20 species of myodocopids have been identified to date from Tampa Bay. The 2 most abundant and frequently occurring species were *Parasterope pollex* and *Rutiderma darbyi* (Table 2). Most taxa occurred in < 1% of the samples and represented < 0.1% of the individuals collected. The Sorensen coefficient showed that *P. pollex* and *Eusarsiella disparalis* were most similar in their co-occurrence (coefficient = 36), followed by *E. texana–Asteropterygion oculitristis* (32) and *E. texana–P. pollex* (31).
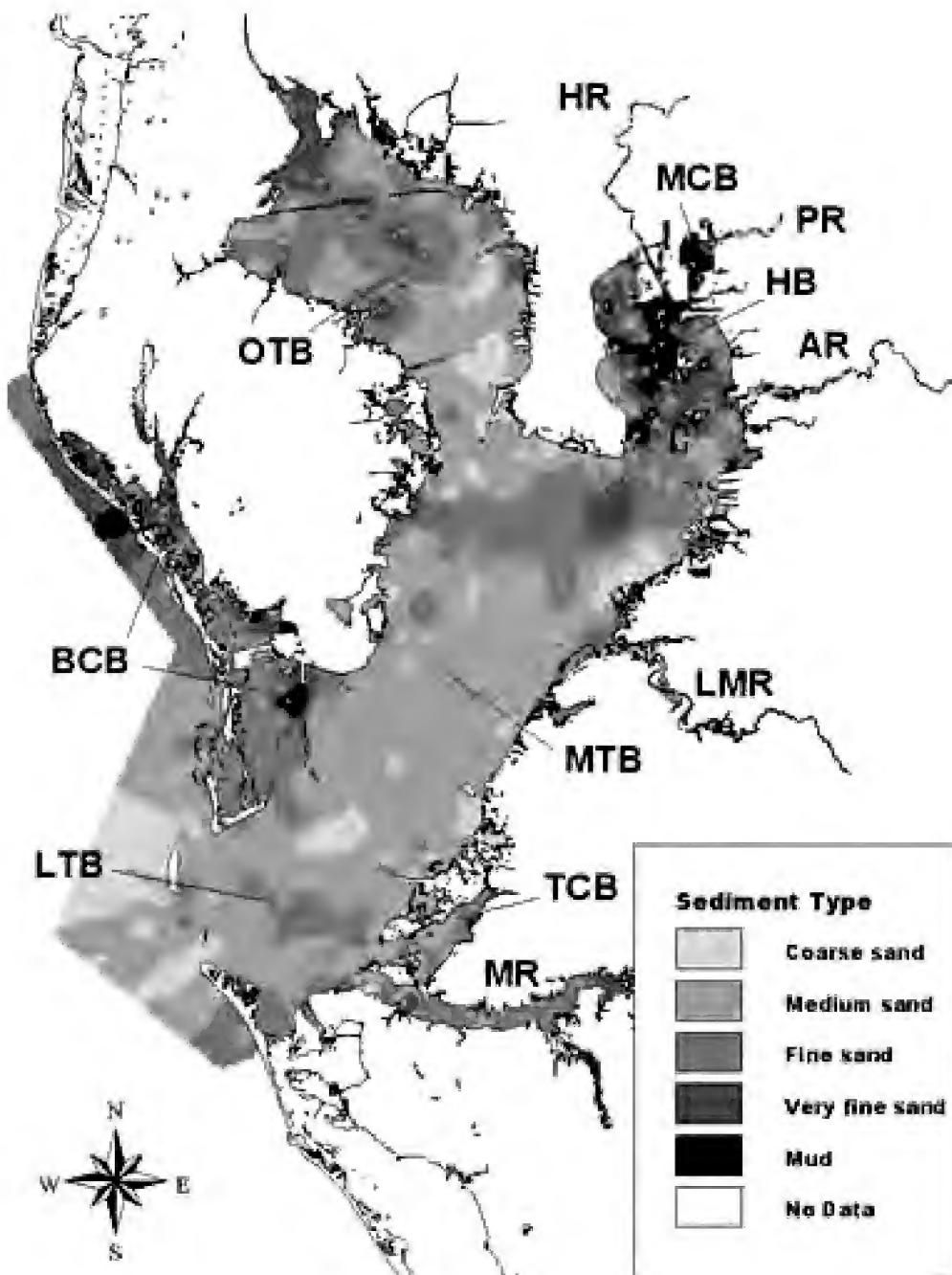
**Figure 2. Map depicting the distribution of sediment types in Tampa Bay, Florida. Site codes are found in Figure 1 legend.**

**Selected taxa**

**Cylindroleberidae.** *Amboleberis americana* has been reported in the Atlantic Ocean from North Carolina to Brazil as well as in the Caribbean and the GOM; it has also been reported from the Pacific coast of central America (Kornicker 1986b). In Tampa Bay, this species was most often found from the central bay to the GOM, generally proximal to the main shipping channel (Figure 3). Logistic regression showed that the probability of occurrence increased with salinity and depth and decreased as %SC increased (Table 3). This species preferred the greatest depths of the 10 species (Table 4). It was also collected over the narrowest range of salinities (Table 4). The optimal sediment type appeared to be coarse sand (%SC < 1.7). A single ovigerous specimen was found with 21 eggs (Table 5); Horsley (1990) reported a maximum of 37 eggs.

*Asteropterygion oculitristis* has been found off coastal Georgia (Darby 1965) and is reported to range to Texas in

TABLE 2

**Taxonomic inventory, frequency of occurrence (%FO), percent composition (%COMP), and mean numbers m$^{-2}$ (standard error (s$_{\bar{x}}$), ±1) of myodocopid ostracods collected from Tampa Bay, Florida, 1995–2001 ($n$ = 610).**

| | % FO | % COMP | Mean (s$_{\bar{x}}$, ±1) # m$^{-2}$ |
|---|---|---|---|
| Cylindroleberidae | | | |
|   *Amboleberis americana* (Müller 1890) | 4.1 | 0.1 | 2.2 (0.6) |
|   *Asteropella* sp. | 0.2 | <0.1 | <0.1 <0.1) |
|   *Asteropella maclaughlinae* Kornicker 1981 | 1.0 | <0.1 | <0.3 (0.1) |
|   *Asteropterygion oculitristis* (Darby 1965) | 12.1 | 0.3 | 6.0 (0.9) |
|   *Parasterope pollex* Kornicker 1967 | 48.2 | 76.4 | 1,621.0 (230.2) |
|   *Prionotoleberis salmoni* Kornicker 1986 | 0.5 | <0.1 | 0.2 (0.1) |
| Philomedidae | | | |
|   *Harbansus paucichelatus* (Kornicker 1958) | 4.6 | 0.2 | 3.9 (1.3) |
|   *Pseudophilomedes ambon* Kornicker 1984 | 0.2 | <0.1 | <0.1 (<0.1) |
|   *Pseudophilomedes darbyi* Kornicker 1989 | 6.2 | 0.2 | 4.5 (1.0) |
| Rutidermatidae | | | |
|   *Rutiderma darbyi* Kornicker 1983 | 28.4 | 19.8 | 420.1 (50.3) |
|   *Rutiderma mollitum* Darby 1965 | 1.8 | 0.6 | 12.2 (6.6) |
| Sarsiellidae | | | |
|   *Eusarsiella* sp. | 1.8 | <0.1 | 0.7 (0.3) |
|   *Eusarsiella childi* Kornicker 1986 | 8.4 | 0.7 | 15.7 (6.8) |
|   *Eusarsiella cresseyi* Kornicker 1986 | 1.6 | <0.1 | 0.5 (0.2) |
|   *Eusarsiella disparalis* (Darby 1965) | 15.7 | 0.6 | 12.2 (2.5) |
|   *Eusarsiella elofsoni* Kornicker 1986 | 0.5 | <0.1 | 0.1 (0.1) |
|   *Eusarsiella ozotothrix* (Kornicker and Bowen 1976) | 0.3 | <0.1 | 0.2 (0.2) |
|   *Eusarsiella radiicosta* (Darby 1965) | 0.8 | <0.1 | 0.5 (0.3) |
|   *Eusarsiella spinosa* (Kornicker and Wise 1962) | 3.4 | 0.1 | 1.3 (0.3) |
|   *Eusarsiella tampa* Kornicker and Grabe 2000 | 2.0 | 0.1 | 1.4 (0.5) |
|   *Eusarsiella texana* (Kornicker and Wise 1962) | 11.8 | 0.8 | 17.4 (7.7) |
|   *Eusarsiella zostericola* (Cushman 1906) | 1.5 | <0.1 | 1.0 (0.5) |
| Family/genera undetermined | 0.8 | <0.1 | 0.2 (0.1) |
| Mean density (Total myodocopid ostracods) | | | 2,121.9 (235.6) |

the GOM (Kornicker 1986b). In Tampa Bay it was primarily found in Middle and Lower Tampa bays (Figure 3). Logistic regression showed that probability of occurrence increased with depth and DO and decreased as %SC increased (Table 3). The optimum habitat appeared to be salinities > 25 psu and sediments of coarse to fine sands (< 8 %SC) (Table 4). In coastal SW Florida, *A. oculitristis* abundance was positively associated with %SC (where %SC ranged up to about 25%) and was negatively associated with the sorting coefficient (Grabe et al. 1995). Brood sizes ranged from 11 to 18 (Table 5).

*Parasterope pollex* has been reported from bays and estuaries from Nova Scotia, Canada, south to the Chesapeake Bay and along the Gulf coast of Florida to depths of about 13 m (Kornicker 1986b, Grabe et al. 1995). *Parasterope pollex* was the most widespread myodocopid

in Tampa Bay (Figure 3) and it was the only species commonly collected in the upper portions of the bay. Bay-wide, *P. pollex* was collected in almost half of the samples, and it was present in 69% of the Old Tampa Bay samples. Densities ranged to 67,350 m$^{-2}$ in Hillsborough Bay and averaged > 3,000 m$^{-2}$ in Old Tampa Bay, 2,500 m$^{-2}$ in Hillsborough Bay and 1,600 m$^{-2}$ bay-wide. This frequency of occurrence and the mean densities are lower than those reported by Hulings (1969) for a bay in Massachusetts. *Parasterope pollex* was found in 90% of his samples from Hadley Harbor (near Martha's Vineyard), and seasonal means for adults ranged from 2,360–,440 m$^{-2}$.

This species was collected in Tampa Bay over the widest ranges of salinity and %SC (Table 3). Grabe et al. (1995) found that *P. pollex* abundance in SW Florida was
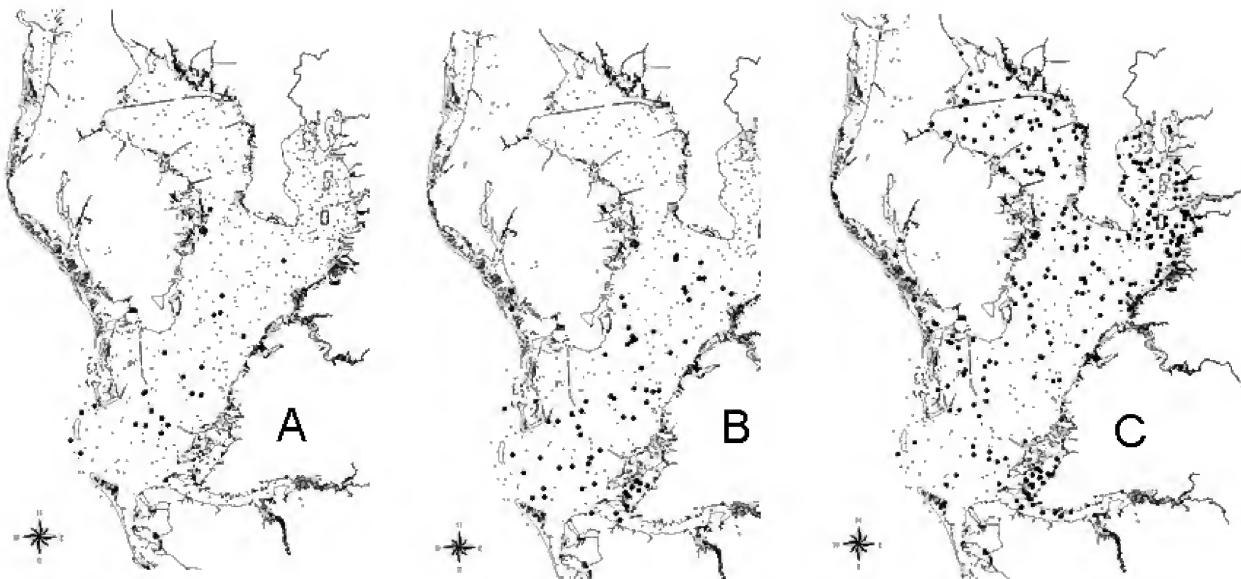
**Figure 3. Distribution of A)** *Amboleberis americana*, **B)** *Asteropterygion oculitristis*, **and C)** *Parasterope pollex* **in Tampa Bay, Florida, 1995–2001.**

associated with fine sand-sized sediments; in this study the optimal sediments were the medium to fine sand-sized sediments that predominate in Tampa Bay (Table 4). *Parasterope pollex* was also the species most tolerant of low DO concentrations (Table 4). Logistic regression showed that depth, DO and %SC were the most important variables explaining the occurrence of *P. pollex* (Table 3). Brood sizes ranged from 4–12 (Table 5). Horsley (1990) summarized data from several studies and estimated that the maximum number of eggs for the largest (1.44 mm CL) female would only be seven.

**Philomedidae.** *Harbansus paucichelatus* is reported to occur from North Carolina into the GOM and to Belize in the Caribbean Sea (Kornicker 1984a). In Tampa Bay, *H. paucichelatus* was found in the middle to lower portions of the bay, including Terra Ceia Bay (Figure 4). Logistic regression showed that salinity and %SC were key variables affecting its occurrence (Table 3). This species preferred coarse to fine sand-sized sediments and polyhaline salinities (Table 4). Horsley (1990) collected this species most often from medium and fine sand-sized sediments. Brood sizes ranged from 3–6 (Table 5).

*Pseudophilomedes darbyi* has been reported from North Carolina south into the GOM as far west as Texas (Kornicker and Iliffe 1989). In Tampa Bay, *P. darbyi* was found mainly in Middle and Lower Tampa Bay but did penetrate into Old Tampa Bay (Figure 4). Logistic regression showed this species' presence to be positively associated with salinity and depth and negatively associated with %SC (Table 3). *Pseudophilomedes darbyi* preferred the

second deepest waters of the 10 species; it prefers the narrowest range of salinities and coarse to medium sands (Table 4). Brood sizes ranged from 3–7 with a median of 4 (Table 5).

**Rutidermatidae.** *Rutiderma darbyi* occurs from North Carolina to south Florida and the Bahamas and into the GOM (Kornicker 1983). It was widespread throughout Middle and Lower Tampa Bay and penetrated midway into both Old Tampa Bay and southern Hillsborough Bay (Figure 4). Logistic regression showed an association with depth, salinity and %SC (Table 3). Although a Gaussian response curve could not be fitted for %SC, the probability of occurrence was 0.7 at 0 %SC (coarse sands) and approached 0 at 15 %SC (very fine sands). Horsley (1990) found *R. darbyi* to be more common in medium and fine sands. Brood sizes were small and ranged from 2–6 eggs (Table 5). Kornicker (1986b) reported that the Rutidermatidae generally brood 3–4 eggs regardless of size.

**Sarsiellidae.** *Eusarsiella childi* was described by Kornicker (1986a) from specimens collected in SW Florida and has been reported from the GOM at depths to 12.8 m. Although *E. childi* was most frequently found in the lower bay, it did penetrate into Old Tampa Bay and Hillsborough Bay; there was also a single occurrence in upper Boca Ciega Bay (Figure 5). Logistic regression showed that %SC and salinity were the key abiotic variables (Table 3). *Eusarsiella childi* appeared to inhabit the narrowest range of sediment types, preferring coarse sands (Table 4). This contrasts with observations off Marco

**TABLE 3**

Summary of forward stepwise logistic regression analyses for the association between selected abiotic variables and the 10 most frequently occurring myodocopid ostracod species.

| | McFadden's Rho$^2$ | Constant | $L_{10}$ Temperature | $L_{10}$ Salinity | $L_{10}$ Depth | $L_{10}$ DO | ASN%SC |
|---|---|---|---|---|---|---|---|
| **Cylindroleberidae** | | | | | | | |
| *Amboleberis americana* | 0.23 | | | | | | |
| Coefficient | | −25.4 | NS | 14.8 | 2.9 | NS | −31.6 |
| Odds Ratio | | | | >2 x 10$^6$ | 19 | | 0 |
| *Asteropterygion oculitristis* | 0.11 | | | | | | |
| Coefficient | | −5.9 | NS | NS | 2.8 | 3.4 | −7.1 |
| Odds Ratio | | | | | 16 | 27 | <1 |
| *Parasterope pollex* | 0.02 | | | | | | |
| Coefficient | | −1.0 | NS | NS | NS | 1.4 | −1.4 |
| Odds Ratio | | | | | 4 | <1 | |
| **Philomedidae** | | | | | | | |
| *Harbansus paucichelatus* | 0.12 | | | | | | |
| Coefficient | | −15.2 | NS | 9.2 | NS | NS | −28.0 |
| Odds Ratio | | | | 9,793 | | | 0 |
| *Pseudophilomedes darbyi* | 0.24 | | | | | | |
| Coefficient | | −7.5 | NS | 4.3 | −3.1 | NS | −25.7 |
| Odds Ratio | | | | 1,039 | 21 | | 0 |
| **Rutidermatidae** | | | | | | | |
| *Rutiderma darbyi* | 0.24 | | | | | | |
| Coefficient | | −7.5 | NS | 4.3 | 3.1 | NS | −25.7 |
| Odds Ratio | | | | 1,039 | 21 | | 0 |
| **Sarsiellidae** | | | | | | | |
| *Eusarsiella childi* | 0.16 | | | | | | |
| Coefficient | | −11.5 | NS | 7.4 | NS | NS | −39.3 |
| Odds Ratio | | | | 1,643 | | | 0 |
| *Eusarsiella disparalis* | 0.02 | | | | | | |
| Coefficient | | −1.3 | NS | NS | NS | NS | −5.2 |
| Odds Ratio | | | | | <0.1 | | |
| *Eusarsiella spinosa* | 0.09 | | | | | | |
| Coefficient | | 34.3 | −26.6 | NS | 2.4 | NS | NS |
| Odds Ratio | | | 0 | | 10 | | |
| *Eusarsiella texana* | 0.04 | | | | | | |
| Coefficient | | 3.8 | −8.2 | 3.1 | NS | 2.1 | NS |
| Odds Ratio | | | 0 | 22 | | 8 | |

Island in SW Florida where *E. childi* abundance was positively associated with %SC (where % SC ranged up to about 25%) (Grabe et al. 1995). *Eusarsiella childi* also tended to occur at shallower depths than many of the other species. Brood sizes ranged from 3–14 (Table 5). Horsley (1990) reported a range of 1–10 eggs in 5 specimens.

*Eusarsiella disparalis* was described from coastal Georgia (Darby 1965) and ranges from North Carolina to just north of Tampa Bay (Kornicker 1986a). *Eusarsiella*

*disparalis* is widespread in Tampa Bay, ranging from the mouth of the bay to the mouths of the Hillsborough and Alafia rivers in Hillsborough Bay (Figure 5). Logistic regression showed that %SC was the most important abiotic variable affecting its presence or absence in Tampa Bay (Table 3). A preferred salinity regime could not be defined using logistic regression. *Eusarsiella disparalis* was collected over the second widest salinity range after *P. pollex*, and was the species that most often co-occurred

**TABLE 4**

Summary of habitat characteristics for the 10 most frequently occurring myodocopid ostracods in Tampa Bay, Florida, 1995-2001. COA = center of abundance (R = range); OPT = optimum (TOL = tolerance). NR = Gaussian logistic regression equation could not resolve either an "optimum" or a "tolerance" range.

| | SALINITY (psu) | %SC | DEPTH (m) | DO (mg/l) |
|---|---|---|---|---|
| **Cylindroleberidae** | | | | |
| *Amboleberis americana* | COA(R): 28.9 (25.3-33.9) | COA(R): 2.8 (0.1-13.2) | COA(R): 4.1 (0.1-12.5) | COA(R): 5.5 (4.3-6.9) |
| | OPT(TOL): 32.0 (27.6-36.4) | OPT(TOL):0 (0-1.4) | OPT(TOL): 12.1 (7.0-17.2) | OPT(TOL): 5.5 (4.5-6.5) |
| *Asteropterygion oculitristis* | COA(R): 28.0 (10.8-35.0) | COA(R): 4.1 (0.1-17.1) | COA(R): 3.9 (0.1-9.0) | COA(R): 5.6 (4.4-8.9) |
| | OPT(TOL): 30.3 (26.2-34.7) | OPT(TOL): 3.3 (0.0-9.3) | OPT(TOL): 7.5 (4.5-10.5) | OPT(TOL): 5.7 (4.9-6.5) |
| *Parasterope pollex* | COA(R): 25.6 (4.3-35.8) | COA(R): 4.3 (0.8-91.8) | COA(R): 2.2 (0.1-11.0) | COA(R): 5.4 (0.2-11.3) |
| | OPT(TOL): 23.6 (17.5-29.7) | OPT(TOL): 7.1 (2.3-12.0) | OPT(TOL): 3.4 (0.7-6.0) | OPT(TOL): 6.3 (2.7-9.9) |
| **Philomedidae** | | | | |
| *Harbansus paucichelatus* | COA(R): 29.0 (20.2-34.5) | COA(R): 3.6 (0.1-6.1) | COA(R): 3.2 (0.1-8.5) | COA(R): 6.2 (4.5-8.9) |
| | OPT(TOL): 29.8 (17.5-29.7) | OPT(TOL): 3.3 (1.0-5.6) | OPT(TOL): 6.7 (3.2-10.2) | OPT(TOL): 5.6 (5.0-6.2) |
| *Pseudophilomedes darbyi* | COA(R): 28.6 (18.0-34.0) | COA(R): 3.6 (0.1-21.4) | COA(R): 5.7 (0.1-12.2) | COA(R): 5.7 (3.4-7.0) |
| | OPT(TOL): 32.6 (32.2-33.0) | OPT(TOL): 3.6 (2.4-4.8) | OPT(TOL): 9.2 (8.5-9.9) | OPT(TOL): 5.8 (5.0-6.6) |
| **Rutidermatidae** | | | | |
| *Rutiderma darbyi* | COA(R): 27.3 (8.2-34.7) | COA(R): 2.9 (0.1-49.6) | COA(R): 4.5 (0.1-12.2) | COA(R): 5.7 (2.5-9.2) |
| | OPT(TOL): 29.2 (24.3-34.0) | OPT(TOL): NR | OPT(TOL): 8.3 (5.0-11.6) | OPT(TOL): 5.9 (4.8-7.0) |
| **Sarsiellidae** | | | | |
| *Eusarsiella childi* | COA(R): 27.4 (17.6-34.7) | COA(R): 2.2 (0.8-6.4) | COA(R): 2.4 (0.1-8.5) | COA(R): 6.6 (3.4-9.2) |
| | OPT(TOL): NR | OPT(TOL): 0.0 (0.0-1.2) | OPT(TOL): 2.9 (1.3-4.5) | OPT(TOL): 7.1 (5.3-8.9) |
| *Eusarsiella disparalis* | COA(R): 25.3 (7.9-36.0) | COA(R): 5.0 (1.4-23.1) | COA(R): 2.1 (0.1-6.5) | COA(R): 5.1 (0.3-9.2) |
| | OPT(TOL): NR | OPT(TOL): 5.0 (3.0-7.0) | OPT(TOL): 2.4 (0.3-4.5) | OPT(TOL): 6.3 (2.8-9.8) |
| *Eusarsiella spinosa* | COA(R): 28.3 (10.8-33.9) | COA(R): 4.2 (0.2-17.1) | COA(R): 4.3 (0.1-9.0) | COA(R): 5.5 (4.1-7.3) |
| | OPT(TOL): 31.9 (27.9-35.9) | OPT(TOL): 0.0 (0.0-3.9) | OPT(TOL): 9.0 (4.8-13.2) | OPT(TOL): 6.6 (5.7-7.5) |
| *Eusarsiella texana* | COA(R): 26.2 (10.8-35.9) | COA(R): 4.2 (0.2-91.8) | COA(R): 2.4 (0.1-9.1) | COA(R): 6.3 (1.6-9.2) |
| | OPT(TOL): NR | OPT(TOL): 5.7 (3.4-8.0) | OPT(TOL): 3.7 (1.2-6.2) | OPT(TOL): 7.2 (5.1-9.3) |
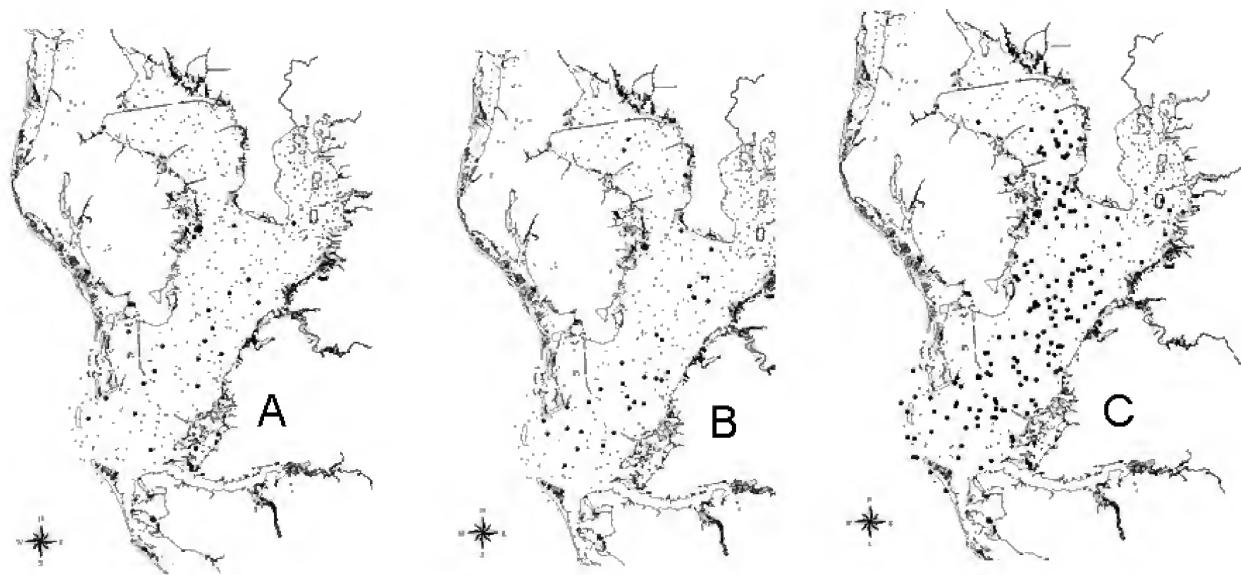
**Figure 4. Distribution of A)** *Harbansus pauchichelatus*, **B)** *Pseudophilomedes darbyi*, **and C)** *Rutiderma darbyi* **in Tampa Bay, Florida, 1995–2001.**

with *P. pollex*. Both *E. disparalis* and *P. pollex* were also the species most tolerant of subnominal (< 4 mg/l) DO (Table 4). The preferred habitat was shallow waters with medium to fine sands (Table 4). In coastal SW Florida *E. disparalis* abundance was positively associated with %SC (where % SC ranged up to about 25%) (Grabe et al. 1995).

Brood sizes ranged from 4–11 (Table 5). Horsley (1990) reported a maximum of 13 in the literature and a range of 1 to 10 in his samples.

*Eusarsiella spinosa* is known to occur from North Carolina to the Indian River on Florida's east coast and in the GOM from Marco Island to Texas (Kornicker 1986a, Grabe et al. 1995). This species was found primarily in the middle and lower portions of Tampa Bay, including Boca Ciega Bay, and Terra Ceia Bay, and was one of only 4 species found in the Manatee River; there was a single occurrence in Hillsborough Bay (Figure 5). Logistic regression showed only a weak association with temperature and depth (Table 3). This species preferred deeper waters and coarse to medium sand-sized sediments (Table 4). Brood sizes were among the smallest of the sarsiellids (Table 5).

*Eusarsiella texana* was described from the Texas Gulf coast (Kornicker and Wise 1962) and has been reported to range to Maryland (Kornicker 1986a). *Eusarsiella texana* is found throughout Tampa Bay, although it did not penetrate deeply into either Hillsborough Bay or the Manatee River (Figure 5). Logistic regression showed that key abiotic variables were salinity, DO, and temperature (Table 3). This species occurred over a wide range of both salinities

and sediment types (Table 4). In coastal SW Florida *E. texana* was most abundant in well-sorted sands and at %SC up to > 25% (the maximum reported in the study, Grabe et al. 1995).

Brood sizes ranged from 2–13 (Table 5) whereas Horsley (1990) reported a maximum brood of 8 eggs.

## TABLE 5

**Mean and range of brood sizes of myodocopid ostracods from Tampa Bay, Florida ($n$ = number of specimens).**

| Species ($n$) | Mean | Range |
|---|---|---|
| Cylindroleberidae | | |
| *Amboleberis americana* (1) | 21 | NA |
| *Asteropterygion oculitristis* (4) | 15 | 11–18 |
| *Parasterope pollex* (30) | 8 | 4–12 |
| Philomedidae | | |
| *Harbansus paucichelatus* (15) | 4 | 2–6 |
| *Pseudophilomedes darbyi* (7) | 5 | 3–10 |
| Rutidermatidae | | |
| *Rutiderma darbyi* (55) | 3 | 2–6 |
| *Rutiderma mollitum* (3) | 3 | 2–4 |
| Sarsiellidae | | |
| *Eusarsiella childi* (45) | 8 | 3–14 |
| *Eusarsiella cresseyi* (7) | 5 | 4–7 |
| *Eusarsiella disparalis* (26) | 7 | 4–11 |
| *Eusarsiella spinosa* (4) | 5 | 2–7 |
| *Eusarsiella tampa* (5) | 6 | 4–8 |
| *Eusarsiella texana* (25) | 6 | 2–12 |
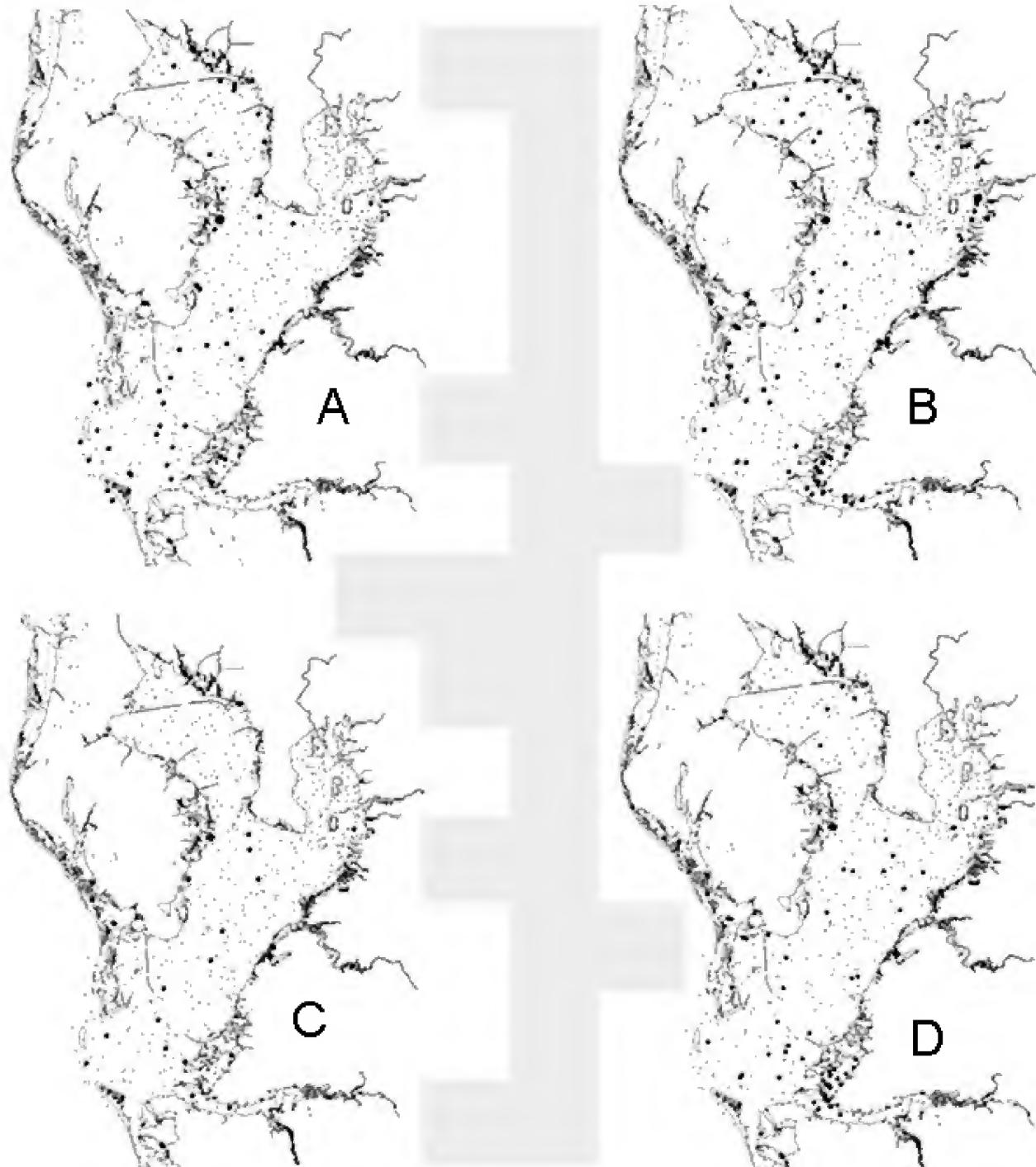
**Figure 5. Distribution of A)** *Eusarsiella childi*, **B)** *E. disparalis*, **C)** *E. spinosa*, **and D)** *E. texana* **in Tampa Bay, Florida,**

## CONCLUSIONS

At least 20 species of myodocopid ostracods were identified from > 600 samples collected from Tampa Bay during 1995–2001. Numerical dominants included *P. pollex* and *R. darbyi*, and the most frequently occurring species included *P. pollex*, *R. darbyi*, and *E. disparalis*. Of the 10 most frequently occurring species, most were more likely to occur in coarser sediments, in more saline waters, and at greater depths than the mean for Tampa Bay. Given that many were described from coastal waters, the abiotic preferences reported here suggest that most of these ostracods penetrate the bay from the GOM. Whether there is a seasonal effect on the distribution is unknown since only wet season samples have been analyzed. However, bay salinities are noticeably higher during the dry season (Lewis and Estevez 1988), and ostracods may be more widespread when salinities are higher. Additionally, the observation that species such as *A. americana* tend to be found near the main shipping channel suggests that enhanced influx of Gulf waters via the shipping channels (Lewis and Estevez 1988) could facilitate immigration of neritic species. In contrast, *P. pollex* is a typical "bay species," less frequently collected in the lower reaches of the bay. Accordingly, *P. pollex* shows greater affinity for lower salinities, finer-grained sediments, and lower DO than the species presumed to originate in the GOM.

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# Gulf and Caribbean Research

Manatee Occurrence in the Northern Gulf of Mexico, West of Florida

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# MANATEE OCCURRENCE IN THE NORTHERN GULF OF MEXICO, WEST OF FLORIDA

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*ABSTRACT* Reports of West Indian manatees (*Trichechus manatus*) in the US Gulf of Mexico west of Florida have increased during the last decade. We reviewed all available manatee sighting, capture, and carcass records (*n* = 377) from Alabama, Louisiana, Mississippi, and Texas since the early 1900s; only 40 of these were previously published. Manatees were reported most often in estuarine habitats, usually either near a freshwater source or natural or industrial warm-water springs/runoffs during winter months. The recent increase in manatee records may be due to a combination of increased public awareness and dispersal of manatees, most likely seasonal migrants from Florida. We caution that the presence of artificial warm-water sources outside of the manatee's traditional range may attract an increasing number of manatees and could increase the incidence of cold-related mortality in this region.

## INTRODUCTION

The West Indian manatee (*Trichechus manatus*) occurs patchily along coastal areas throughout the Gulf of Mexico (GOM) and Caribbean Sea, the southeastern United States, and the northern and eastern waters of South America (Lefebvre et al. 2001). Historically, manatees were found along the entire GOM Coast from the Suwannee River in Florida to the Bay of Campeche, Mexico, and considered common in south Texas (Gunter 1941, Powell and Rathbun 1984, Lefebvre et al. 2001). Records of manatee sightings, carcasses, and captures west of Florida are limited, but have increased in recent years.

## MATERIALS AND METHODS

All available historical and current (up to August 2004) sighting, carcass, and capture records (excluding archeological data) of manatees in the northern GOM, west of Florida were compiled for this report. Our goal was to provide a comprehensive document that included records collected from numerous widely scattered resources. Powell and Rathbun (1984) and Rathbun et al.

(1990) provided the most recent reviews of manatee records in this area. Resources used to provide new data included marine mammal stranding networks in each state west of Florida, current literature, as well as files of the Sirenia Project (US Geological Survey), Fish and Wildlife Research Institute (Florida Fish and Wildlife Conservation Commission [FWC]), and the US Fish and Wildlife Service [USFWS]. Since the occurrence data were compiled from a variety of sources, the precision and accuracy of the data also varied. We corrected any errors noted in published accounts. When only geographic descriptions were given, we determined coordinates as near as possible to the physical description. The locations with geographic coordinates were not assumed to be correct and were verified. If coordinates did not fit the description, the record was verified for accuracy and then moved as close to the original geographic description as possible. The October 2001 sighting of a manatee 144.8 km south of Mobile Bay in open water was excluded from Appendix I and any tabulations, since it was not attributable to any state waters. This sighting is included in the map, and information is presented in the discussion. Manatees are physically and behaviorally distinctive from all other marine mammals.

Therefore, species identification by the general public could usually be confirmed with a general description. Confirming multiple sightings of the same manatee or different animals is problematic, and unless a manatee has distinguishing marks (e.g., propeller scars), it is impossible to identify single individuals (Beck and Reid 1995).

## RESULTS

We compiled 377 records from US waters west of Florida; these were 339 sightings, 30 carcasses, and 8 captures. Sighting and carcass records are presented in Appendices I and II, while captures are presented in the state summaries. Forty of the 377 records were previously published. Louisiana ($n = 147$) and Alabama ($n = 132$) accounted for the majority of the occurrence records (39% and 35%, respectively). All sighting, carcass, and capture records are plotted in Figure 1 along with major waterways and intermittent wetland habitat which serve as possible transit pathways for manatees.

### Alabama

Alabama's records consisted of 128 sightings, 4 carcasses, and no captures (Appendices I, II, Figure 1). Two of these records were previously published. The years with the most records were 1995 (20 sightings), 2000 (23 sightings and 1 carcass), and 2002 (22 sightings and 1 carcass). June had the most information ($n = 29$). Sighting size varied from 1 to 11 manatees; single individuals were most frequent. Cow/calf pairs (including multiple pairs) made up 14 sightings (Appendix I). An occurrence record noted by Caldwell and Caldwell (1973) near Gulf Shores was excluded since the type (sighting or carcass) was not noted.

The Alabama Marine Mammal Stranding Network (AMMSN) received reports of a lone manatee on 13, 16, and 25 December 1991. On 13 January 1992, an adult male manatee was found dead on the east shore of Mobile Bay (Appendix II, AMMSN number SHCM 119). There was no immediate, obvious indication as to the cause of its death; however, the animal had large and round fecal obstructions in the intestine consistent with possible exposure to cold. In fact, water temperature in the bay recorded two weeks earlier was 11 °C. This assessment is consistent with description of cold stress syndrome described by Buergelt et al. (1984) and Bossart et al. (2003). Due to the proximity in sighting dates with the actual carcass recovery, this was likely the same individual sighted during December 1991 (Appendix I).

On 26 February 2002, a 260 cm dead male manatee was found in Mobile County, on the south shore of Dog River (Appendix II, AMMSN SHCM 350). A field necropsy revealed that the entire intestinal tract was black inside, and the large intestine contained solidified masses resembling charcoal briquettes. The cause of death was cold stress, again consistent with the descriptions provided in Buergelt et al. (1984) and Bossart et al. (2003).

On 11 September 2003, 11 manatees were sighted in McReynolds Lake at the Mobile-Tensaw River Delta. Two of the 11 were matched to manatees known from the Crystal River population in northwest Florida; both were males, one known since 1982, and the other since 1987 (Sirenia Project files).

### Mississippi

Mississippi's records consisted of 27 sightings, 3 carcasses, and two captures (Appendices I, II, Figure 1). Ten of these were previously published. The greatest number of sightings was recorded in 1979 (5 sightings and one capture) and 2001 (4 sightings and 1 carcass). February ($n = 7$) and August ($n = 6$) had the most records. Single individuals were most frequently sighted. Two of the three compiled carcass records indicated that the animals died from starvation and/or cold stress.

Powell and Rathbun (1984) reported one carcass and 24 manatee sightings in Mississippi between 1978 and 1981; 16 of these sightings occurred near Biloxi Bay between 28 November 1979 and 19 January 1980, but no further details were provided. A 1.8 m individual was caught in a trawl and released alive on 3 December 1981 in Graveline Bayou (30.33333, –88.66666) (Gunter and Perry 1983, Powell and Rathbun 1984). One male manatee, "Beauregard," was rescued by Sea World of Florida in January 1979 at Gulfport Harbor (30.35000, –89.16667) and relocated to Florida. He was rehabilitated in captivity and released in February 1985. Using a satellite tag, Sirenia Project biologists tracked him upon release (Gunter and Corcoran 1981, Powell and Rathbun 1984, Rathbun et al. 1990). "Beauregard" was tracked from the Homosassa River to the Suwannee River in Florida (Rathbun et al. 1990). Powell and Rathbun (1984) noted that Gunter & Corcoran (1981) erroneously reported the capture date to be 7 January 1979.

### Louisiana

Louisiana's records consisted of 131 sightings, 15 carcasses, and one capture (Appendices I, II, Figure 1). Eight of these were previously published. The years with the most records were 1995 (23 sightings and 3 carcasses) and 2002 (24 sightings and 2 carcasses). Eighty-nine percent ($n = 130$) of the 147 records provided seasonal information; June and July had the most records with 21 and 31,
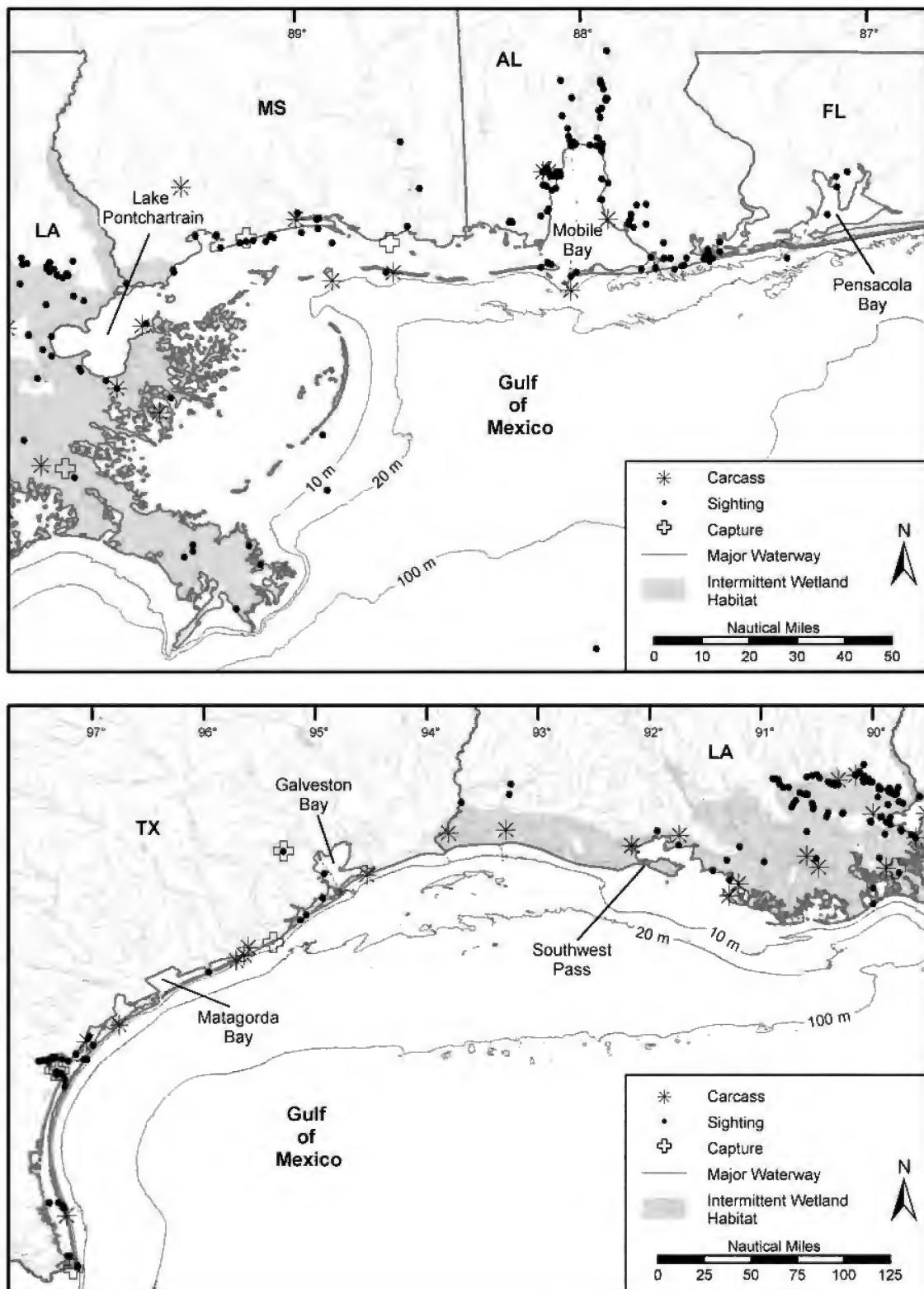
Figure 1. Distribution of manatee records (*n* = 378) from 1853–2004 in the northern Gulf of Mexico, west of Florida. Maps are separated, east and west of the Mississippi River delta, for visual clarity and resolution. Major waterways and intermittent wetland habitats (shaded areas lacking resolution to depict small waterways) are depicted; these demonstrate potential pathways of manatee movement.

respectively. Single individuals were most frequently sighted, though occasional sightings of cow/calf pairs were made (Appendix I).

On 22 July 1995, a single manatee was sighted about 4.8 km southeast of Breton Sound Marina in a canal near Hopedale, Louisiana. This individual was later unintentionally hit and killed by an oil well crew boat (> 25 m in length) (Appendix II). The carcass (LA9501) was collected by the Louisiana Department of Wildlife and Fisheries, photographed, buried, and later recovered by the Louisiana Marine Mammal Stranding Network. Photographs were subsequently matched to an animal previously photographed in Tampa Bay, Florida, in February 1995 (Anonymous 1996).

Another individual was seen repeatedly for several weeks in November 1995 in the 21 °C warm-water effluent of the Michoud Power Plant (Appendix I). On 31 January 1996, following a sharp drop in air temperature, a dead manatee was observed floating out of a waste-water discharge pipe on the south shore of the lake (Appendix II); this was probably the same animal sighted in November. The necropsy revealed that the animal had been feeding up to the time of death. The cause of death was determined to be entrapment in the discharge pipe and subsequent drowning (J. Valade, personal communication, US Fish and Wildlife Service, 6620 Southpoint Drive South, Suite 310, Jacksonville, FL 32216-0958).

A manatee photographed on 10 August 1999 in Bayou Lacombe was later matched to a carcass (no assigned specimen number) recovered in Bayou Patout on 14 December 1999 (Appendix II). Assuming travel along the complex coastline of Louisiana, this is a distance of about 417 km, and included crossing the mouth of the Mississippi River delta. It is possible that this individual made its way through the intricate bayou system of Louisiana, though this is speculative.

On 8 September 1999, a 3.3 m individual was caught in a trawl and released alive west of the Mississippi River near West Pointe a la Hache (29.54322, –89.80227).

### Texas

Texas' records consisted of 53 sightings, 8 carcasses, and 5 captures (Appendices I, II, Figure 1). Twenty-one of the 66 records were previously published. The most records were from 1995 (20 sightings and 1 capture) and 2001 (12 sightings and 1 carcass). Eighty-six percent ($n$ = 57) of the 66 records provided seasonal information; October and November had the most records with 14 and 12, respectively. Single individuals were most frequently sighted, and there were repeated sightings of a cow/calf pair in the Galveston Bay area in 1995.

Six of the 21 published records were live captures from the southern Texas coast (Laguna Madre and Rio Grande) (Gunter 1941). Manatee sightings listed in Table 1 of Powell and Rathbun (1984) are in error, in that True (1884) mentions only one manatee for the GOM Coast. It was Gunter (1941) that is the correct source for a number of sightings (depending on interpretation, about 8 sightings, with some captures) in southern Texas. It appears that the numbers for those two source documents were transposed. The captures took place during 1853–1855 at Brazos, but sources did not specify Brazos Island or Brazos Santiago Pass. Specimen USNM 1375 at the US National Museum, Washington, DC, is one of those individuals.

Between 3 and 8 September 1995, a manatee cow/calf pair was sighted feeding on unidentified seagrasses in west Galveston Bay (Appendix I). There were repeated reports of these animals being sighted within 3 km of this location for the next week (Schiro and Fertl 1995). On 15 September 1995, the pair was seen near the west end of Galveston Island (Appendix I) (about 9 km east of San Luis Pass). Fishermen also sighted a cow/calf pair near North Deer Island on 18 September 1995 (Appendix I). All of these sightings were likely of the same pair.

On 25 October 1995, a manatee was sighted at the Barney Davis Power Plant located on the Laguna Madre near the town of Flower Bluff. A second sighting was made 31 October 1995 at the Naval Air Station at Ingleside. On the morning of 2 November 1995, a manatee was observed throughout the day, several kilometers away at the Rockport Harbor and boat basin. Estimated body length was 305 cm, and the individual was determined by ventral observation to be a female. A notch on the right side of the tail, white marks above both eyes and a barnacle behind the right eye were noted. On 6 November 1995, a manatee with the same markings was swimming in a debris-strewn drainage ditch at the Koch Refinery on the La Quinta Channel, Corpus Christi. The manatee later moved into the La Quinta Channel heading towards the Central Power and Light plant. Water temperature at that time was about 20 °C. On 8 November 1995, the same manatee was sighted and videotaped near the Texas State Aquarium (Appendix I). The final sighting was on 12 November 1995 in Port Aransas at the University of Texas Marine Science Institute (UTMSI) boat basin. The manatee remained in the basin throughout the day feeding on turtle grass (*Thalassia testudinum*), shoal grass (*Halodule wrightii*), and mangrove seeds. A scrape mark was observed behind the eye where the barnacle had been. The manatee was last observed and videotaped near dark at the far end of the Port Aransas Municipal Harbor (Appen-

dix I). This was the last known sighting of this individual. Throughout late November and early December 1995, a single manatee was repeatedly sighted in Buffalo Bayou, just west of downtown Houston. This individual was most often observed at the warm-water outfall of a municipal wastewater treatment plant. On occasion, however, the manatee was seen leaving the canal, moving into the Houston Ship Channel. On one occasion, the individual was sighted 16 km downstream by a tow boat captain but was resighted the next day in its original location (Russel 1996). USFWS and Texas Parks and Wildlife Department (TPWD) personnel captured the manatee on 7 December 1995. The female manatee (313 cm) was moved to Sea World of Texas in San Antonio for temporary holding and nicknamed "Sweetpea." Genetic analysis determined that she was a Florida manatee (R. Bonde, personal communication, US Geological Survey, Florida Integrated Science Center, Sirenia Project, 412 NE 16th Avenue, Room 250, Gainesville, FL 32601, Garcia-Rodriguez et al. 1998). "Sweetpea" was later transferred to Sea World in Orlando, Florida and spent the winter rehabilitating at Homosassa Springs State Wildlife Park in Citrus County, Florida (Weigle et al. 2001). She was satellite-tagged and released at the headwaters of the Homosassa River on 23 April 1996 (Weigle et al. 2001). Once released, she swam northward along the west coast to the Florida Panhandle, spending most of the spring and summer at sites around Apalachee Bay (Weigle et al. 2001). After moving west to Apalachicola Bay in September, she reversed her course and began heading south along the west coast, visiting various locations before reaching Marathon in the Florida Keys in November 1996 (Weigle et al. 2001). "Sweetpea" then took a northeast turn along the Florida Keys and wintered in south Miami. "Sweetpea's" tag stopped transmitting in mid-March 1997 in Brevard County on Florida's central east coast. Her last known location was where the Banana River joins the Indian River (Weigle et al. 2001). Her entire tag assembly, including belt, was recovered on 17 March 1999 in the Indian River, just south of Sebastian Creek (middle of Atlantic Coast of Florida) (Sirenia Project files). The belt had been cut (possibly by a propeller; however, this was not confirmed). She has not been sighted since. We believe that the manatee seen in late October and early November 1995 in Corpus Christi could be "Sweetpea", based on fluke notches, similar size, and same sex, but confirmation is not possible. On 14 July 2001, TPWD personnel sighted a manatee in the Rockport area (Appendix I). During the last week of July 2001, a manatee was spotted in the UTMSI boat basin in Port Aransas (Appendix I). On 11 September 2001 a manatee estimated to be about 2.13 m in length was sighted in the

Hampton's Landing Boat Basin in Aransas Pass (Appendix I). On 23 September 2001, a manatee, estimated to be about 1.83 m in length, was sighted in the inlet between the Texas State Aquarium and the Lexington Museum in Corpus Christi, Texas (Beaver 2001). On 3 October 2001, a manatee was videotaped near the Texas State Aquarium. Scars were observed on the left dorsum of the animal; however, the photograph quality was too poor to attempt a match to any known individual using identifying marks. This manatee spent time around the dock at the aquarium. Observers were able to determine that the individual was a male. A manatee was seen 11 Oct and 26 Oct near Portland (Appendix I), roughly 9.66 km from the aquarium. On 14 November 2001, a manatee was spotted at Valero Refining Company in Corpus Christi (Appendix I). When last seen, it was heading west towards Koch Refinery and the end of the Tule Lake Channel. On 29 November 2001, the manatee appeared emaciated to on-site biologists. A rescue attempt was initiated on 30 November 2001, but personnel from USFWS, Sea World, and UTMSI were unsuccessful in attempts to locate the manatee on 30 November 2001. The sightings from 29 and 30 November were in the inner harbor, where there are some warm-water outfalls. A manatee was seen again 5 and 12 December near Portland, in the same area as the 11 October 2001 sighting. Each of the reported sightings in November and December indicated that the manatee was becoming more lethargic and emaciated. On 12 December 2001, a cold front hit the area and dropped the air temperatures to about 7 °C. Repeated trips to the area where the manatee had been sighted yielded no further sightings of the individual.

During this same time period in October 2001, another individual was found dead and floating at Sargent Beach (Matagorda County), just off the Intracoastal Waterway, 241.4 km west of Port Aransas (Appendix II). The water temperature was about 23 °C. This manatee was a male, 3.05 m in length, and contrary to the editor's note associated with Beaver (2001), this could not be the same individual as reported above in the Port Aransas area. A tissue sample was collected from this individual and submitted for genetic analysis. This specimen matched the Florida manatee haplotype (R. Bonde, personal communication, Garcia-Rodriguez et al. 1998).

Most recently, from late June to mid-August 2004, there were several sightings of manatees in south Texas (Appendix I). Seven sightings of one or perhaps even 2 individuals were reported in the area of Port Aransas and Corpus Christi Bay.

## DISCUSSION

Manatees occurring west of Florida and to the north of Mexico generally are considered to be strays originating from populations in either Florida or Mexico (e.g., Gunter 1941, Lowery 1974, Powell and Rathbun 1984, Domning and Hayek 1986). Many manatees in Florida make seasonal movements northward in spring and southward in the fall (Moore 1951a, Powell and Rathbun 1984). Coinciding with these movements, manatees in Mexico move north into Tamaulipas (near the US/Mexico border) during the rainy season (May through September) (Lazcano-Barrero and Packard 1989). The most likely source of emigrants along the GOM coast would be manatees that over-winter in the headwaters of the Crystal and Homosassa rivers, as well as perhaps the Tampa–Ft. Myers region (Bonde and Lefebvre 2001). This is supported by the photographic matches made to manatees sighted in Alabama and Louisiana, as well as genetic analyses of tissue samples from two individuals found in Texas.

Researchers have documented wide-ranging movements by some West Indian manatees. Data for some individuals in Florida suggest a traditional long-range seasonal migration along the Atlantic coast (Reid et al. 1991, Deutsch et al. 2003). Annual movements in excess of 1,700 km (round trip) have been documented for one radio-tagged manatee on the Atlantic coast. "Chessie" moved between Florida and the Chesapeake Bay in multiple years and one year migrated as far as Rhode Island (Deutsch et al. 2003). "Gina," a manatee photo-identified as a calf and juvenile in the Homosassa River on the GOM coast of Florida, has been living in the Bahamas since about 1996 (Reid 2000, Lefebvre et al. 2001). A manatee hit by a crew boat in Louisiana was photo-identified in the Tampa Bay area (Anonymous 1996, FWC files), a minimum coastal distance of 618 km. Two manatees were sighted in the Dry Tortugas in 1982 (Reynolds and Ferguson 1984), and a wayward manatee radio-tagged at Crystal River in northwest Florida was rescued just six weeks later off the Dry Tortugas in 1998 (Sirenia Project files). Hartman (1979) also mentioned sightings of manatees in the Dry Tortugas. The impetus for wide ranging movements is not always apparent but is likely in response to environmental cues; for males, it may be a strategy for mate-searching as well (Deutsch et al. 2003).

We found manatees to be most common in estuarine and river mouth habitats and rare in the open ocean. This observation mirrors their natural history, although data collection is heavily skewed to coastal observations. This habitat preference has been noted by other sources (Moore 1951b, Hartman 1979, Rathbun et al. 1982). Occasionally

manatees may wander into deep waters. Schwartz (1995) commented on the rare occurrence of open ocean sightings off North Carolina. A manatee was sighted about 12.87 km off the Louisiana coast in early July 1979 (Gunter and Corcoran 1981). More recently, a manatee was sighted on 15 October 2001 about 144.8 km south of Mobile Bay, in waters over the Mississippi Canyon in Minerals Management Service's Lease Block Mississippi Canyon 85 during oil and gas exploration operations (Anonymous 2001; T. Pitchford, personal communication, Florida Fish and Wildlife Conservation Commission, Marine Mammal Pathobiology Laboratory, 3700 54th Avenue, South, St. Petersburg, FL 33711; Sirenia Project files). The exact location of the manatee was not recorded, but the center coordinates for this 3 square mile block are −87.94482, 28.91394, with a bottom depth greater than 1,524 m (Sirenia Project files), not 914.4 m as reported by Anonymous (2001). The manatee was sighted for a few days around operating vessels and was even observed to feed on algae growth on the bottom of the vessel. Efforts were underway to attempt a rescue, but the manatee disappeared when several large sharks were seen in the vicinity. The manatee was last sighted on 17 October 2001, and its fate remains unknown.

During the warm season, adult males are considered to range over wider areas than females and subadults (Bengtson 1981, Deutsch et al. 2003). Based on five manatees captured or stranded in South Carolina and Georgia, Rathbun et al. (1982) suggested that extralimital animals would mostly be males. Information on the age or sex for most of the individuals in this review was not available; however, we were able to determine that all age and sex classes appear to make extended range movements. Interestingly, 7% of all the occurrence records were of cow/calf pairs. Deutsch et al. (2003) found that subadults in the Atlantic subpopulation demonstrated strong philopatry to specific warm-season ranges that they had occupied as calves, and some followed the same migratory patterns as their mothers.

Access to warm water, freshwater, and food is required by manatees (Hartman 1979). Temperature is the overriding factor in determining the geographic extent of suitable habitat to manatees (Smith 1993). The vulnerability of manatees to cool ambient water temperatures is well-documented (Moore 1951b, O'Shea et al. 1985, Miculka and Worthy 1995). Manatee deaths attributed to exposure to cold were recorded as early as the 19th century (Moore 1951b, O'Shea et al. 1985, Ackerman et al. 1995). Data suggest that manatees possess metabolic rates that are only 25–30% of predicted values (Gallivan and Best 1980, Irvine 1983, Miculka and Worthy 1995), resulting in a lack

of cold tolerance. Young manatees (< 300 kg) are even more susceptible to cold than adults because they are apparently incapable of increasing metabolic rate at low temperatures (Miculka and Worthy 1995), possibly resulting in hypothermia and death. To offset these metabolic insufficiencies, manatees respond to cold weather by relocating to thermal refuges, either natural spring or warm-water industrial effluents. As noted by Moore (1951b), large springs have immense flow averages that can supply water at 22 °C much faster than the air can chill it. Mothers introduce their offspring to warm-water refuges during the prolonged period of dependence common to the species (Hartman 1979, Deutsch et al. 2003). This suggests the possibility that in the future there may be increased dependence on warm-water sites along the northern GOM. We observed signs similar to those described as cold-stress in many of the manatees found dead west of Florida. Several of the winter sightings were at natural warm springs and industrial warm-water effluents.

Residents near some warm-water springs in Alabama report regularly seeing manatees over the past 40 years. They consider these sightings unremarkable. There are probably other localized areas along the northern GOM coast where forage is available and water temperatures might be high enough and consistently reliable to support manatees through the winter. For example, manatees have been seen near power plant and wastewater treatment plant effluents in both Louisiana and Texas, particularly during winter months. Additionally, the USFWS (2001) noted that canals and boat basins, where warmer water temperatures persist as temperatures in adjacent bays and rivers decline, might also be used as temporary thermal refuges. Manatees in this study were often observed in such habitats.

Gunter (1941) reported that all manatees observed in Texas at the time were seen during the summer months and that manatee presence would be precluded in any part of Texas during midwinter. Powell and Rathbun (1984) suggested that sightings have declined in frequency and that all have occurred during the summer. While there were many records for summer, we noted a considerable number of more recent winter sightings as well. These individuals concentrated their movements in boat basins and at power plant effluents. In addition, the public is more aware of the sensitivity of manatees to cold than in the past.

Access to freshwater also influences the movements of manatees. Manatees are attracted to freshwater from natural sources such as rivers and springs, as well as from anthropogenic sources such as wastewater or storm-water outfalls, drainage pipes, and garden hoses (O'Shea and Kochman 1990, Lefebvre et al. 2001, Weigle et al. 2001). Osmoregulatory studies demonstrate that while manatees

can cope with brackish water environments, they cannot survive prolonged exposure to the marine environment unless they can visit freshwater sources on a regular basis (Ortiz et al. 1998).

Seagrasses are a main component of a manatee's diet in coastal areas (Lefebvre et al. 2000); although Florida manatees are generalists, feeding on a wide variety of aquatic vegetation, emergent or terrestrial vegetation, algae, grass trimmings from mowing, and fish carcasses (e.g., Powell 1978, Smith 1993, Baugh et al. 1999, Lefebvre et al. 2000, 2001). Some seagrass-associated invertebrates may be incidentally consumed during foraging on vegetation (e.g., Mignucci and Beck 1998); however, they may also be preferentially ingested (Courbis and Worthy 2003). Lefebvre et al. (2000) suggested that Florida manatees benefit the most by eating available forage in proximity to their refuges or travel routes. Seagrass beds of *Thalassia* and *Halodule* are more extensive from Mobile Bay to Florida Bay than in the rest of the GOM (Handley 1995). These seagrasses west of Mobile Bay exist only in isolated patches and in narrow bands to Aransas Bay, Texas (Handley 1995). Freshwater submerged aquatic vegetation also occurs throughout GOM estuaries and river deltas (Handley 1995). Manatee grass (*Syringodium filiforme*) and shoal grass (*Halodule wrightii*) are the dominant seagrasses found in the shallow water on the northern side of the barrier islands of Mississippi (Handley 1995). Coastal Louisiana has a large amount of submerged aquatic vegetation, with only a small portion of this being seagrasses (Handley 1995). The only remaining seagrass beds in coastal Louisiana exist in Chandeleur Sound (Handley 1995). There is a wide distribution of seagrasses, predominantly shoal grass and widgeon grass (*Ruppia maritima*), in the Galveston Bay estuary (Handley 1995). Seagrasses are prevalent in Laguna Madre (Onuf 1995). Seagrass meadows are increasing in upper Laguna Madre; however, they are on the decrease in lower Laguna Madre (Onuf 1995). There are small patches of shoal grass and widgeon grass (*Ruppia maritima*) in the Corpus Christi Bay area (McCullough 2001, Pulich et al. 1997) and patches of red turf algae (*Gelidium* spp.) and sea lettuce (*Ulva* spp.) (L. Price-May, personal observation).

We compiled various reports of manatees feeding west of Florida. One manatee in Port Aransas, Texas was observed to feed on loose sea grasses such as turtle grass (*Thalassia testudinum*), shoal grass, cordgrass (*Spartina* spp.), mangrove seeds, and other vegetable material. A manatee cow-calf pair was seen feeding on seagrasses (unidentified species) in Galveston Bay, Texas. Additionally, one manatee sighted in the Natalbany River

(Louisiana) was feeding on lilies (unidentified species), a second was sighted in Lake Maurepas (Louisiana) in a Hydrilla (*Hydrilla* spp.) bed, while another in open water off the southwest tip of the Chandeleur Islands was feeding on a weed line at the water's surface.

There is evidence that manatees can be temporarily independent of warm water, perhaps moving to nearby seagrass beds to feed (Bengtson 1981, Shane 1984, Deutsch et al. 2003). Some of the animals reported in the present study in the vicinity of New Orleans, Houston, and Port Aransas (described in detail earlier) were often observed leaving warm-water refuges, only to return several hours later, perhaps having consumed food. Periodic movements from wintering sites at Blue Springs, Florida, and at power plants have been noted (Bengtson 1981, Irvine 1983, Deutsch et al. 2003). As suggested by Smith (1993), it is probable that manatees may leave warm-water areas only after air and adjacent water temperatures have risen in the afternoon and only after cold fronts have passed. Several Alabama manatees were sighted in warm-water refuges without food resources; however, nearby waters could supply an abundance of food. Irvine (1983) noted that manatees would leave warm-water refuges to feed in cooler waters only if they can shortly return to the warmer water temperatures to digest their food.

Traveling manatees use warm-water refuges along their migratory routes during both the early spring and late fall in a 'stepping-stone' strategy, which may permit them to migrate earlier in the spring as well as remain at sites later into the fall (Reid et al. 1991, Deutsch et al. 2003). Individuals may disperse during intervening periods of mild weather with warmer temperatures (Moore 1956, Hartman 1979, Shane 1984, Reid et al. 1991).

Numerous sightings, for example in lakes St. Catherine and Pontchartrain in Louisiana, northern Mobile Bay in Alabama, and Corpus Christi Bay/Laguna Madre in Texas, suggest repeated use of certain areas. Individual manatees in Florida and Georgia are known to return to the same winter ranges each year, and some may also return to the same summer ranges (Rathbun et al. 1982, 1990, Koelsch 1997, Deutsch et al. 2003). Seasonal site fidelity has also been noted for some radio-tagged manatees frequenting southeastern Georgia (Zoodsma 1991). It is not known whether the manatees mentioned in this paper were the same individuals returning annually to the same area. More attempts to photo-identify these strays would provide additional information. Studies also should be conducted to characterize the habitat in these areas to determine what might attract individuals and ensure proper management strategies.

The reasons are not known for the large number of extralimital sightings of this species along the GOM. Collard et al. (1976) noted that as the health of northern GOM estuaries and their associated flora improves, the excursion range of manatees may broaden. Bonde and Lefebvre (2001) suggested that the increase in sightings might have been made possible by man-made sources of warm waters (such as industrial effluents), as well as a decade of relatively warm winters. Storm events and a climatic trend of warmer winters and summers may also help to explain increased extralimital movements by manatees (Lefebvre et al. 2001). In Texas and Louisiana, we noted a peak in 1995 of the number of manatee sightings west of Florida. The 1995 hurricane season was a notably active one for major storms, with 19 named storms (the mean is nine), 11 of which became hurricanes (the mean is five) (Williams and Duedall 1997). It was not a record but a close second to the 1933 season of 21 storms (Williams and Duedall 1997). Langtimm and Beck (2003) determined significant annual variation in adult manatee survival in years when intense hurricanes and a major winter storm occurred in the northern GOM. Many of the manatee sightings we compiled for west of Florida occurred after four hurricanes and three tropical storms entered the GOM in 1995; several of these storms directly impacted Florida and the Yucatan Peninsula. As noted by Langtimm and Beck (2003), a storm might cause manatees to emigrate from Florida either voluntarily (in response to cooled surface waters which follow in the wake of a hurricane and can persist for days) or involuntarily (e.g., by strong longshore currents or high-energy waves). For example, a manatee was sighted in Theodore Channel in Alabama during Hurricane Opal in October 1995. The growing public awareness of the manatee also may be a sufficient explanation for the increased number of reports (Rathbun et al. 1982, Schwartz 1995, Lefebvre et al. 2001, Schleifstein 2004). Lastly, the increase in extralimital sightings west of Florida is probably due to animals moving from the southern Big Bend coast, where their numbers have increased (Rathbun et al. 1990, Bonde and Lefebvre 2001).

From this review, it is obvious that small numbers of manatees occasionally migrate through the northern GOM from Florida and possibly Mexico. Because of these movement patterns, environmental planners and managers need to consider the likelihood that manatees may be affected by a variety of human activities in coastal waters (as well as deeper waters, on occasion) of the northern GOM. Increased attention also must be given to the protection of habitat resources throughout the manatee's travel corridors (Smith 1993). For example, Handley (1995) notes that losses of seagrasses in the northern GOM have been exten-

sive, varying 20–100% for most estuaries. As in Florida, alterations to both natural and industrial warm-water refuges along the rest of the GOM coast have significant implications for manatees (USFWS 2001). For example, human activities in the vicinity of these springs and the use of aquifer waters are a threat to the availability and suitability of spring waters to manatees. If the volume of water flowing from springs decreases, available and accessible habitat and water temperature around springs may drop, increasing manatees' exposure to cold waters and its associated health risks. The status of manatees as an endangered species makes the loss of individual manatees biologically significant. We hope that this compilation will stimulate further investigations of manatee distribution west of Florida in the northern GOM and serve as continued encouragement for people to report occurrences of manatees to appropriate personnel. To that end, it is requested that future manatee observations be reported to the appropriate authorities in each state and to the USFWS' Jacksonville Field Office, which is charged with the daily management of the Florida manatee and holds the recovery lead for the species. A secure, electronic database is maintained to record and track all manatee sightings, rescues, and deaths outside the state of Florida. To contribute data to the manatee sighting and stranding network contact the USFWS office at 904-232-2580, extension 123 to receive a username and password. A yearly summary for all out-of-state manatee activity is sent to all manatee stranding network partners.

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# APPENDIX I

**Sighting records of manatees in the northern Gulf of Mexico, west of Florida.** c/c = cow/calf pair

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|------|---------------------|----------|-----------|--------------|--------|
| | **ALABAMA** | | | | |
| 13 Jun 1985 | Tensaw River, 25 mi south of Jackson, Baldwin County | 31.00329 | -87.90777 | 1 | O'Shea (1988), Rathbun et al. (1990) |
| 21 Oct 1986 | Bon Secour Bay, 0.5 mi east of Highway 59 bridge | 30.28000 | -87.68333 | 1 | O'Shea (1988)[1] |
| 13 Dec 1991 | Fish River, app. 1.6 km north of Highway 98 | 30.43283 | -87.82217 | 1 | unpub. data |
| 16 Dec 1991 | Fish River at Highway 98 | 30.41467 | -87.82500 | 1 | unpub. data |
| 25 Dec 1991 | Theodore Channel turning basin | 30.53283 | -88.11883 | 1 | unpub. data |
| 31 May 1993 | 1 mi west of Perdido Pass | 30.27833 | -87.56783 | 1 | unpub. data |
| 31 May 1993 | Gulf State Park Resort, south beach, Gulf side | 30.25350 | -87.64467 | 3 | unpub. data |
| 1 Jun 1993 | lagoon at Bon Secour National Wildlife Refuge | 30.24133 | -87.78550 | 1 | unpub. data |
| 29 Sep 1993 | Mobile River | 30.73500 | -88.04400 | 1 | unpub. data |
| 29 Oct 1994 | Theodore Channel turning basin | 30.53133 | -88.11883 | 1 | unpub. data |
| 31 Oct 1994 | north side of Dauphin Island | 30.25833 | -88.10000 | 1 | unpub. data |
| 4 Nov 1994 | west Mobile Bay, north of Dog River | 30.58167 | -88.07217 | 1 | unpub. data |
| 8 Nov 1994 | Dog River, Alligator Bayou | 30.56867 | -88.10250 | 1 | unpub. data |
| 9 Nov 1994 | Intracoastal Waterway | 30.27883 | -87.68333 | 1 | unpub. data |
| 11 Nov 1994 | Dog River, Grand Mariner Marina | 30.58217 | -88.08500 | 1 | unpub. data |
| 11 Nov 1994 | Dog River, Rabbit Creek | 30.58217 | -88.13033 | 1 | unpub. data |
| 14 Nov 1994 | Dog River, mouth of Halls Mill Creek | 30.59667 | -88.11467 | 1 | unpub. data |
| 15 Nov 1994 | Blakely River, south of I-10 | 30.65417 | -87.92550 | 1 | unpub. data |
| 16 Nov 1994 | Fowl River, near Bellingrath Gardens | 30.43200 | -87.13717 | 1 | unpub. data |
| 18 Nov 1994 | Mobile Bay, west side near I-10 | 30.68783 | -88.02450 | 1 | unpub. data |
| 26 Nov 1994 | Mobile Bay, near I-10 | 30.68867 | -88.03533 | 1 | unpub. data |
| 6 Dec 1994 | Dog River, Rabbit Creek | 30.59033 | -88.12117 | 1 | unpub. data |
| 1 Apr 1995 | Mobile Delta, McReynolds Lake | 30.88967 | -87.92467 | 1 | unpub. data |
| 3 Apr 1995 | Perdido Bay, Terry Cove | 30.29167 | -87.55083 | 1 | unpub. data |
| 9 Apr 1995 | Perdido Bay, Orange Beach | 30.27917 | -87.55467 | 1 | unpub. data |
| 10 Apr 1995 | Perdido Bay, Terry Cove | 30.29167 | -87.55083 | 1 | unpub. data |
| 24 May 1995 | Tensaw River | 30.79717 | -87.93400 | 1 | unpub. data |
| 29 May 1995 | Mobile River, Cedar Creek near Mt. Vernon | 31.05565 | -87.99986 | 1 | unpub. data |
| 18 Jun 1995 | Fowl River, west of Highway 163 | 30.44217 | -88.11550 | 2(c/c) | unpub. data |

[1]O'Shea (1988) reported Hwy 57, but the correct location is Hwy 59 (Sirenia Project files)

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**APPENDIX 1 (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|------|---------------------|----------|-----------|--------------|--------|
| | **ALABAMA (continued)** | | | | |
| week of 25 Jun 1995 | mouth of Fowl River, western shore side | 30.45133 | -88.10817 | 2 (c/c) | unpub. data |
| 4 Jul 1995 | Tensaw River, Hurricane Bayou | 30.83750 | -87.90467 | 1 | unpub. data |
| 14 Sep 1995 | Tensaw River, south end of Gravine Island | 30.77283 | -87.92933 | 4 | unpub. data |
| 24 Sep 1995 | Tensaw River, south end of Gravine Island | 30.77283 | -87.92950 | 1 | unpub. data |
| 25 Sep 1995 | Mobile Delta, east side, north of Highway 90 | 30.67833 | -87.92133 | 1 | unpub. data |
| 4 Oct 1995 | Theodore Channel, turning basin | 30.53167 | -88.11833 | 1 | unpub. data |
| 21 Oct 1995 | Apalachee River, near Blakely River | 30.69917 | -87.94000 | 1 | unpub. data |
| 6 Nov 1995 | Mobile River, Alabama State Docks | 30.70667 | -88.04117 | 1 | unpub. data |
| 14 Nov 1995 | Fish River, Farragut Cove | 30.46550 | -87.80300 | 1 | unpub. data |
| 16 Nov 1995 | Fish River, Farragut Cove | 30.46550 | -87.80300 | 1 | unpub. data |
| 18 Nov 1995 | Fish River, Farragut Cove | 30.46550 | -87.80300 | 1 | unpub. data |
| 21 Nov 1995 | Fish River, Farragut Cove | 30.46550 | -87.80300 | 1 | unpub. data |
| 7 Dec 1995 | Fish River at Hwy. 98 | 30.41467 | -87.82467 | 1 | unpub. data |
| 3 Apr 1996 | Mobile Bay near Dog River | 30.56633 | -88.08550 | 1 | unpub. data |
| 16 Apr 1996 | Gulf of Mexico, Gulf Shores | 30.25450 | -87.63717 | 1 | unpub. data |
| 29 Apr 1996 | Gulf of Mexico, south of Fort Morgan | 30.21717 | -88.03083 | 1 | unpub. data |
| 2 Jul 1996 | Tensaw River, south end of Hurricane Bayou | 30.83333 | -87.91083 | 1 | unpub. data |
| 3 Jul 1996 | Tensaw River, The Basin, Smith Creek | 30.87083 | -87.91850 | 1 | unpub. data |
| 20 Aug 1996 | Theodore Ship Channel | 30.53333 | -88.12717 | 1 | unpub. data |
| 29 Aug 1996 | Tensaw River, The Basin | 30.80550 | -87.92383 | 2 | unpub. data |
| 19 Sep 1996 | Mobile Bay, west side and North of Dog River | 30.58083 | -88.07183 | 2 | unpub. data |
| 22 Sep 1996 | Tensaw River, near railroad bridge | 30.83883 | -87.91083 | 2 | unpub. data |
| 10 Jul 1997 | Mobile Bay, 0.75 mi east of Ft. Morgan | 30.23167 | -88.01333 | 1 | unpub. data |
| 10 Aug 1997 | Dog River, mouth of Rabbit Creek | 30.58833 | -88.12167 | 1 | unpub. data |
| 15 Aug 1997 | Mobile Bay, mouth of Fly Creek | 30.54167 | -87.90167 | 1 | unpub. data |
| 8 Sep 1997 | Mobile Bay, 0.5 mi north of Dog River | 30.56800 | -88.06717 | 1 | unpub. data |
| 25 May 1998 | Mobile Bay, mouth of Pinto Pass | 30.67767 | -88.01767 | 1 | unpub. data |
| 18 Jul 1998 | Mobile Bay, north edge Hwy 90 | 30.67583 | -87.96667 | 1 | unpub. data |
| 24 Oct 1998 | Mobile Bay, north edge Hwy 90 | 30.67583 | -87.96667 | 3(c/c) | unpub. data |
| 24 May 1999 | south side of Dauphin Island | 30.24717 | -88.13583 | 1 | unpub. data |
| 9 Oct 1999 | confluence of Tensaw and Blakeley Rivers | 30.69767 | -87.93583 | 5(c/c) | unpub. data |
| 25 Apr 2000 | Perdido Bay, east of Bear Point | 30.30150 | -87.51250 | 1 | unpub. data |
| 15 May 2000 | Theodore Ship Canal | 30.53333 | -88.12967 | 1 | unpub. data |
| 4 Jun 2000 | head of Bayou La Batre | 30.40417 | -88.23767 | 2(c/c) | unpub. data |

**APPENDIX I (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **ALABAMA (continued)** | | | | |
| 11 Jun 2000 | Intracoastal Waterway, in connector canal from Mobile Bay to Perdido Bay | 30.27917 | -87.70633 | 1 | unpub. data |
| 11 Jun 2000 | Intracoastal Waterway, west of Highway 59 | 30.27867 | -87.70883 | 1 | unpub. data |
| 17 Jun 2000 | South Fork Deer River upstream from confluence with Theodore Ship Canal | 30.51717 | -88.09833 | 1 | unpub. data |
| 17 Jun 2000 | South Deer River, near Theodore Ship Canal | 30.52250 | -88.08417 | 1 | unpub. data |
| 22 Jun 2000 | Theodore Ship Canal | 30.52250 | -88.08417 | 1 | unpub. data |
| 22 Jun 2000 | Dog River, near Alligator Bayou | 30.56667 | -88.09283 | 1 | unpub. data |
| 22 Jun 2000 | Dog River, near mouth to Mobile Bay, near Texaco Dock | 30.56667 | -88.09017 | 1 | unpub. data |
| 23 Jun 2000 | Fish River | 30.43167 | -87.82283 | 1 | unpub. data |
| 25 Jun 2000 | Confederate Bay, north side of Dauphin Island, near marina, east of bridge to mainland | 30.26500 | -88.11250 | 3(c/c) | unpub. data |
| 25 Jun 2000 | Dauphin Island Bay, Hudson Bay, near marina, east of bridge to mainland | 30.26133 | -88.11183 | 3(c/c) | unpub. data |
| 29 Jun 2000 | McReynolds Lake Basin | 30.89433 | -87.92883 | 6–8(c/c) | unpub. data |
| 5 Jul 2000 | Bayou Sara off Mobile River | 30.84050 | -88.03000 | 1 | unpub. data |
| 6 Jul 2000 | Bayou La Batre | 30.40533 | -88.24833 | 1 | unpub. data |
| 9 Jul 2000 | Bayou La Batre | 30.41050 | -88.24333 | 1 | unpub. data |
| 14 Jul 2000 | Dog River | 30.59083 | -88.11300 | 3 | unpub. data |
| 14 Jul 2000 | Dog River | 30.58950 | -88.12250 | 3 | unpub. data |
| 14 Aug 2000 | Bon Secour River | 30.33200 | -87.70783 | 5 | unpub. data |
| 3 Oct 2000 | confluence of Tensaw and Blakeley Rivers | 30.69767 | -87.93583 | 5(2 c/c?) | unpub. data |
| 13 Oct 2000 | Under a fishing pier, Gulf of Mexico | 30.24250 | -87.66867 | 1 | unpub. data |
| 28 Nov 2000 | mouth of Chickasaw Creek | 30.77883 | -88.06117 | 1 | unpub. data |
| 25 Feb 2001 | Intracoastal Waterway | 30.27867 | -87.70867 | 1 | unpub. data |
| 30 Apr 2001 | The Basin | 30.89083 | -87.92583 | 2(c/c) | unpub. data |
| 2 Aug 2001 | Weeks Bay | 30.40000 | -87.83200 | 1 | unpub. data |
| 13 Aug 2001 | west side of Mobile Bay at McNally Park | 30.90117 | -88.06833 | 4 | unpub. data |
| 26 Aug 2001 | Blakely River, Marker 2 | 30.66283 | -87.92367 | 1 | unpub. data |
| 29 Aug 2001 | Blakely River, north of Hwy 90 | 30.66950 | -87.92500 | 1 | unpub. data |
| 31 Aug 2001 | east shore of Mobile Bay at Point Clear | 30.48500 | -87.92550 | 4 | unpub. data |
| 17 Dec 2001 | Fowl River near Bellingrath Gardens | 30.42633 | -88.13583 | 1 | unpub. data |
| 17 Jan 2002 | Magnolia River headwaters | 30.40000 | -87.76833 | 1 | unpub. data |
| 26 Feb 2002 | south bank of Dog River | 30.58117 | -88.10967 | 1 | unpub. data |

**APPENDIX 1 (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **ALABAMA (continued)** | | | | |
| 1 Apr 2002 | Perdido Pass area | 30.28033 | -87.54833 | 1 | unpub. data |
| 17 Apr 2002 | Perdido Bay, Old River, under bridge to Ono Island | 30.28033 | -87.53617 | 1 | unpub. data |
| 18 Apr 2002 | Perdido Bay, Old River by Ono Island | 30.28033 | -87.53617 | 1 | unpub. data |
| 27 Apr 2002 | The Basin, Maple Creek | 30.90200 | -87.92950 | 1 | unpub. data |
| 12 Jun 2002 | Intracoastal Waterway | 30.30583 | -87.55367 | 1 | unpub. data |
| 14 Jun 2002 | Little Lagoon Pass | 30.27050 | -87.54967 | 1 | unpub. data |
| 15 Jun 2002 | east shore of Mobile Bay at Point Clear | 30.48500 | -87.92617 | 2(c/c) | unpub. data |
| 16 Jun 2002 | Intracoastal Waterway | 30.27967 | -87.27767 | 1 | unpub. data |
| 17 Jun 2002 | Bay La Launch at Intracoastal Waterway marker #72 | 30.30972 | -87.55333 | 1 | unpub. data |
| 28 Jun 2002 | The Basin, Maple Creek | 30.90200 | -87.92967 | 1 | unpub. data |
| 30 Jun 2002 | Little Lagoon Pass | 30.27050 | -87.54967 | 1 | unpub. data |
| 2 Jul 2002 | Intracoastal Waterway | 30.28283 | -87.75383 | 1 | unpub. data |
| 6 Jul 2002 | Alabama Point | 30.27967 | -87.62833 | 1 | unpub. data |
| 24 Jul 2002 | Theodore Industrial Canal | 30.52917 | -87.10467 | 1 | unpub. data |
| 19 Aug 2002 | mouth of the Dog River | 30.56533 | -87.10700 | 1 | unpub. data |
| 27 Aug 2002 | Little Lagoon | 30.24550 | -87.73717 | 1 | unpub. data |
| 29 Aug 2002 | Buccaneer Yacht Club | 30.58167 | -87.06783 | 4 | unpub. data |
| 30 Aug 2002 | Buccaneer Yacht Club | 30.58167 | -87.06783 | 4 | unpub. data |
| 4 Oct 2002 | Pinto Pass. Mobile County, just south of *USS Alabama* | 30.67861 | -88.01667 | 4(2 c/c?) | unpub. data |
| 28 Oct 2002 | Intracoastal Waterway | 30.38800 | -87.81333 | 1 | unpub. data |
| 25 May 2003 | Little Lagoon, Baldwin County | 30.24639 | -87.73722 | 1 | unpub. data |
| 1 Aug 2003 | just west of the pass from Gulf into Perdido Bay | 30.27972 | -87.55250 | 1 | unpub. data |
| 11 Sep 2003 | McReynolds Lake, Mobile–Tensaw River Delta | 30.67229 | -88.03221 | 11 | unpub. data |
| 30 Sep 2003 | Bayou La Batre | 30.40500 | -88.24833 | 6(2 c/c) | unpub. data |
| 5 May 2004 | Perdido Pass | 30.33436 | -87.51311 | 1 | unpub. data |
| 7 May 2004 | Terry Cove, near Perdido Bay | 30.28810 | -87.54920 | 1 | unpub. data |
| 24 May 2004 | Intracoastal Waterway near Highway 59, in canal | 30.27917 | -87.68278 | 1 | unpub. data |
| 5 Jun 2004 | McReynolds Lake | 30.55639 | -87.92472 | 1 | unpub. data |
| 8 Jun 2004 | delta north of Mobile Bay, Tensaw River slough | 30.67183 | -87.94333 | 1 | unpub. data |
| 12 Jun 2004 | Fish River | 30.41600 | -87.82367 | 1 | unpub. data |
| 15 Jun 2004 | Perdido Bay, north side of west tip of Perdido Key | 30.28038 | -87.54557 | 1 | unpub. data |
| 18 Jun 2004 | Perdido Bay, in canal leading to Orange Beach Marina | 30.28625 | -87.56365 | 1–2 | unpub. data |
| 30 Jun 2004 | Fish River near Farragut Cove | 30.46883 | -87.76933 | 2(c/c) | unpub. data |

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| **ALABAMA (continued)** | | | | | |
| 24 Jul 2004 | west side of Mobile Bay at McNally Park | 30.58033 | -88.07200 | 5(2 adults; 3 calves) | unpub. data |
| 26 Jul 2004 | canal that is tributary to Alligator Bayou (tributary to Dog River) | 30.56600 | -88.10500 | 1 | unpub. data |
| **MISSISSIPPI** | | | | | |
| 31 Dec 1978 | Wolf River, near the Cueres Bridge | 30.35779 | -89.27396 | 1 | Powell and Rathbun (1984) |
| 1 Jan 1979 | Wolf River, near the Cueres Bridge | 30.35870 | -89.27216 | 1 | Powell and Rathbun (1984) |
| 3 Jan 1979 | Gulfport Coast Guard Station | 30.36000 | -89.08170 | 1 | Powell and Rathbun (1984) |
| 5 Jan 1979 | Gulfport Harbor | 30.35506 | -89.08552 | 1 | Powell and Rathbun (1984) |
| 18 Jan 1979 | Pascagoula River, 0.5 mi from Mississippi Sound | 30.52316 | -88.56308 | 1 | Gunter and Corcoran (1981), Powell and Rathbun (1984) |
| 4 Jul 1979 | 12 mi east of Breton Island, Louisiana | 29.46667 | -88.88333 | 1 | Gunter and Corcoran (1981), Powell and Rathbun (1984)[2] |
| 28 Nov 1979– 19 Jan 1980 | Biloxi Bay | 30.41751 | -88.91401 | 1 | Powell and Rathbun (1984) |
| Summer 1992 | Pearl River, near railroad bridge closest to Mississippi Sound | 30.19205 | -89.58450 | 1 | unpub. data |
| Summer 1992 | Jourdan River, north end of St. Louis Bay | 30.35879 | -89.34588 | 1 | unpub. data |
| 26 Nov 1992 | Mississippi Power's Plant Watson, in intake canal, Gulfport | 30.33186 | -89.18880 | 1 | unpub. data |
| 16 Feb 1993 | Mississippi Sound, near rock jetty adjacent to Buccaneer State Park (near Waveland) | 30.34020 | -89.14210 | 1 | unpub. data |
| 10 Aug 1994 | Bay St. Louis. Bayou Caddy fishing dock | 30.23839 | -89.42107 | 1 | unpub. data |
| 13 Aug 1994 | just off the Long Beach Harbor Jetty Pier (west of Gulfport) | 30.33747 | -89.16674 | 1 | unpub. data |
| 20 Aug 1994 | Hidden Creek Bayou, near power plant (north of Biloxi) | 30.36854 | -88.97535 | 2 | unpub. data |
| 15 Jun 1996 | Gulfport, Whitehouse Sand Bar, 1 mi offshore | 30.33333 | -88.86667 | 1 | unpub. data |
| early Aug 1997 | Front Beach, Biloxi | 30.38333 | -88.91667 | 1 | unpub. data |
| 19 Aug 1997 | Back Bay, near mouth of Biloxi River, by Mississippi River | 30.43684 | -88.98744 | 1 | unpub. data |
| 27 Aug 1997 | West Pascagoula River, Gautier-Hickory Hills area north of the mouth of Bluff Creek, Jackson County | 30.68689 | -88.62812 | 1 | unpub. data |

[2]Powell and Rathbun (1984) reported this record as being west, and not east, of Breton Island contrary to Gunter and Corcoran (1981)

APPENDIX 1 (continued)

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|------|---------------------|----------|-----------|--------------|--------|
| **MISSISSIPPI (continued)** | | | | | |
| 8 Feb 2001 | Horn Island | 30.23000 | -88.68000 | 1 | unpub. data |
| 7 Jul 2001 | Moses Pier, Gulfport Harbor | 30.35364 | -89.07972 | 1 | unpub. data |
| 30 Oct 2001 | Gulfport small craft harbor, next to Mississippi Aquarium | 30.33714 | -89.09977 | 1 | unpub. data |
| early Nov 2001 | Gulfport Harbor | 30.35311 | -89.07246 | 1 | unpub. data |
| 17 Feb 2002 | Harrison, Biloxi | 30.41706 | -88.92133 | 1 | unpub. data |
| 20 Feb 2002 | Pass Christian, Mallini Bayou, app. 300 yards from St. Louis Bay | 30.31637 | -89.25769 | 1 | unpub. data |
| 22 Jun 2002 | Vancleave, along sandbar in Pascagoula River in 5 ft of water | 30.39138 | -88.60314 | 1 | unpub. data |
| 18 May 2003 | Wolf River, 1 mi south of I-10 in Harrison County | 30.36067 | -89.27067 | 1 | unpub. data |
| 18 Dec 2003 | Bayou Caddy near the Mississippi/Louisiana border | 30.23051 | -89.41935 | 1 | unpub. data |
| **LOUISIANA** | | | | | |
| 17 May 1943 | Lake St. Catherine near Grand Point | 30.13215 | -89.73512 | 1 | Lowery (1974), Gunter and Corcoran (1981) |
| 8 Apr 1975 | Norco Shell Oil Plant | 30.01103 | -90.41778 | 1 | Powell and Rathbun (1984) |
| 10 Jul 1976 | Atchafalaya River Swamp, 7 km SW of Morgan City | 29.62077 | -91.31142 | 1 | Powell and Rathbun (1984) |
| 22 Apr 1985 | Amite River, 0.5 mi north of confluence of Bayou Manchac | 30.35172 | -90.90090 | 1 | O'Shea (1988), Rathbun et al. (1990) |
| 29 Apr 1985 | Blind River, I-10 highway bridge | 30.12668 | -90.71951 | 1 | O'Shea (1988), Rathbun et al. (1990) |
| 11 May 1985 | Pass Manchac, 3 mi west of Lake Pontchartrain | 30.29727 | -90.35292 | 1 | O'Shea (1988), Rathbun et al. (1990) |
| 21 May 1985 | Pass Manchac, 1 mi west of Lake Pontchartrain | 30.29995 | -90.34176 | 1 | O'Shea (1988), Rathbun et al. (1990) |
| 22 Jun 1986 | Ascension Parish, Amite River canal, 3 km upstream from Port Vincent | 30.34849 | -90.87216 | 1 | unpub. data |
| 9 Sep 1988 | Tchefuncta River at Marina Del Rey Condos near Madisonville | 30.41703 | -90.12505 | 1 | unpub. data |
| Oct or Nov 1989 | mouth of Tchefuncta River | 30.37705 | -90.16102 | 1 | unpub. data |
| 21 Apr 1990 | pond adjacent to canal at Chevron Plant, Venice | 29.23610 | -89.38390 | 1 | unpub. data |
| 13 Oct 1990 | Amite River just above Port Vincent bridge | 30.34510 | -90.85293 | 1 | unpub. data |
| 14 Jun 1991 | Intracoastal Canal in Homa/Morgan City area | 29.60421 | -90.98066 | 1 | unpub. data |
| 18 Jul 1991 | Amite River near Diversion Canal and Blind River | 30.27642 | -90.81250 | 1 | unpub. data |
| 21 Jul 1991 | Caernarvon Canal | 29.85870 | -89.89867 | 1 | unpub. data |
| Mar 1992 | Bonnet Carre Spillway | 30.04706 | -90.40687 | 2 | unpub. data |
| 6–12 Nov 1993 | Eden Isles development, North Shore | 30.22053 | -89.79255 | 1 | unpub. data |

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | LOUISIANA (continued) | | | | |
| 27 May 1994 | Grand Isle, Three Oaks Center | 29.23491 | -90.00534 | 1 | unpub. data |
| 16 May 1995 | Oil rig slip off Blind River, 0.25 mi from Lake Maurepas | 30.21507 | -90.60213 | 1 | unpub. data |
| 17 May 1995 | at abandoned marina in Bayou Liberty near Slidell, 1 mi up bayou from Lake Pontchartrain | 30.26269 | -89.85345 | 1 | unpub. data |
| Jun 1995 | North Pass at Manchac | 30.32116 | -90.40715 | 1 | unpub. data |
| 15 Jun 1995 | North Pass just west of Middle Bayou | 30.31381 | -90.35476 | 1 | unpub. data |
| 17 Jun 1995 | North Pass and Pass Manchac | 30.30554 | -90.32442 | 1 | unpub. data |
| 17 Jun 1995 | Pass Manchac, just west of Lake Pontchartrain | 30.30002 | -90.31339 | 1 | unpub. data |
| early Jul 1995 | US Coast Guard Station, Venice | 29.25556 | -89.35236 | 1 | unpub. data |
| 10 Jul 1995 | North Pass and Middle Bayou | 30.30278 | -90.33453 | 1 | unpub. data |
| 10 Jul 1995 | Lake Pontchartrain, 3 mi out in lake from Bayou Lacombe | 30.19021 | -89.95863 | 2(c/c) | unpub. data |
| 22 Jul 1995 | App. 3 miles southeast of Breton Sound Marina in canal. near Hopedale. St. Bernard Parish | 29.82267 | -89.61985 | 1 | unpub. data |
| 26 Jul 1995 | North Pass at Manchac | 30.30700 | -90.38800 | 1 | unpub. data |
| 7 Oct 1995 | Retainer Canal adjacent to Mississippi/ Gulf outlet between Violet Canal and Bayou Bienvenue Canal | 29.95889 | -89.87937 | 1 | unpub. data |
| 13 Oct 1995 | Reserve Relief Canal near Airline Hwy | 30.08447 | -90.54424 | 1 | unpub. data |
| 29 Oct 1995 | Reserve Relief Canal just west of LaPlace between Airline Hwy and Lake Maurepas | 30.10275 | -90.54152 | 1 | unpub. data |
| Nov 1995 | Reserve Relief Canal just west of LaPlace between Airline Highway and Lake Maurepas | 30.10275 | -90.54152 | 1 | unpub. data |
| 12 Nov 1995 | Eden Isles | 30.21869 | -89.79004 | 1 | unpub. data |
| 12 Nov 1995 | Michoud Power Plant canal on the Mississippi River | 30.01312 | -89.92717 | 1 | unpub. data |
| 17 Nov 1995 | Michoud Power Plant canal on the Mississippi River | 30.01312 | -89.92717 | 1 | unpub. data |
| 18 Nov 1995 | Michoud Power Plant canal on the Mississippi River | 30.01312 | -89.92717 | 1 | unpub. data |
| 5 Dec 1995 | In Lake Pontchartrain 2 mi from Chef Pass heading toward the Rigolets, near Irish Bayou, app. 5 mi offshore | 30.14867 | -89.76949 | 1 | unpub. data |
| 14 Dec 1995 | Eden Isles development near Slidell | 30.21869 | -89.79004 | 1 | unpub. data |
| mid-Dec 1995 | Michoud Power Plant | 30.01000 | -89.92900 | 1 | unpub. data |
| 18 Dec 1995 | Tickfaw River at Hwy 22, Killian | 30.37724 | -90.54779 | 1 | unpub. data |
| 9 Jul 1996 | Eden Isles development near Slidell | 30.21869 | -89.79004 | 1 | unpub. data |
| 10 Jul 1996 | Lake Borgne | 30.05000 | -89.51500 | 1 | unpub. data |
| 28 Sep 1996 | Lake Borgne at mouth of Bayou Bienvenue | 30.00577 | -89.84812 | 1 | unpub. data |
| 25 Jun 1997 | Mouth of Alligator Bayou, Blind River | 30.22709 | -90.65445 | 1 | unpub. data |

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **LOUISIANA (continued)** | | | | |
| 26 Jun 1997 | Mouth of Alligator Bayou, Blind River | 30.22709 | -90.65445 | 1 | unpub. data |
| 1 Jul 1997 | Bourgeois Canal off Blind River just north of I-10 | 30.15497 | -90.69688 | 1 | unpub. data |
| late Jul 1997 | Amite River | 30.30000 | -90.84000 | 1 | unpub. data |
| late Jul 1997 | "The Weir" Amite at Diversion Canal | 30.33480 | -90.85830 | 1 | unpub. data |
| 9 Aug 1997 | Lake Maurepas at mouth of Blind River | 30.21295 | -90.58940 | 1–2 | unpub. data |
| 12 Oct 1997 | confluence of Natalbany and Tickfaw Rivers | 30.36069 | -90.48620 | 1 | unpub. data |
| mid-Jun 1998 | mouth of Blind River near Lake Maurepas | 30.21220 | -90.59441 | 1 | unpub. data |
| 29 Jun 1998 | In Jefferson/St. Charles Canal heading from Lake Pontchartrain into swamp | 30.03625 | -90.27769 | 3 | unpub. data |
| March thru mid-Jul 1998 | Blind River near the mouth | 30.21082 | -90.60991 | 1 | unpub. data |
| 9 Aug 1998 | Bayou Lacombe near gas pipeline company seaway (near seaplane base) | 30.28128 | -89.95272 | 1 | unpub. data |
| 29 Oct 1998 | passing through lock on Inner Harbor Navigational Canal near New Orleans | 29.96582 | -90.02349 | 1 | unpub. data |
| 22 Nov 1998 | near mouth of Bayou Liberty/Bayou Bonfouca near Lake Pontchartrain | 30.24937 | -89.86533 | 1 | unpub. data |
| Jan 1999 | Atchafalaya Delta | 29.44681 | -91.29490 | 1 | unpub. data |
| 13 Jun 1999 | near where Violet Canal meets Lake Borgne (near Martello Castle) | 29.93743 | -89.84788 | 1–2 | unpub. data |
| 25 Jun– 19 Jul 1999 | north shore of Lake Pontchartrain. Mandeville Boat Harbor | 30.37083 | -90.08308 | 2(c/c) | unpub. data |
| 10 Aug 1999 | Flat Lake, near Morgan City | 29.74359 | -91.20041 | 1 | unpub. data |
| Summer 1999 | Blind River near Airline Hwy, (within 3 mi of St. James Boat Club) | 30.09553 | -90.72939 | 1 | unpub. data |
| Summer 1999 | around Mandeville harbor | 30.34856 | -90.06206 | > 1 | unpub. data |
| Summer 1999 | around the Causeway at Lake Pontchartrain, Mandeville | 30.36242 | -90.08878 | > 1 | unpub. data |
| Summer 1999 | Green Point, off Fontainebleau State Park in Lake Pontchartrain | 30.32364 | -90.04163 | > 1 | unpub. data |
| 10 Aug 1999 | Bayou Lacombe | 30.25833 | -89.95000 | > 1 | unpub. data |
| 1 Sep 1999 | High Bridge Canal between Madisonville and the lighthouse | 30.38475 | -90.16550 | 1 | unpub. data |
| 10 Sep 1999 | Tchefuncte River, St. Tammany Parish | 30.38793 | -90.15710 | 1 | unpub. data |
| 8 Nov 1999 | Sabine River just north of I-10 bridge | 30.14023 | -93.69567 | 1 | unpub. data |

**APPENDIX I (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **LOUISIANA (continued)** | | | | |
| 21–24 Nov 1999 | west fork of Calcasieu River and Calcasieu River, Lake Charles | 30.29794 | -93.24755 | 1 | unpub. data |
| 28 Nov 1999 | boat slip off salt mine canal on southeast end of Avery Island | 29.88327 | -91.93877 | 1 | unpub. data |
| 5 Dec 1999 | Bayou Patout. northeast of Weeks Island. west of junction of Patout Canal & Iberia/St. Mary Canal | 29.75671 | -91.74669 | 1 | unpub. data |
| 7 Dec 1999 | between Socola Canal #1 and Foster's Canal in Grand Bayou | 29.51090 | -89.76691 | 1 | unpub. data |
| 10 Dec 1999 | between Socola Canal #1 and Foster's Canal in Grand Bayou | 29.51090 | -89.76691 | 1 | unpub. data |
| 21 Apr 2000 | Martello Castle, Lake Bourne | 29.88517 | -89.74491 | 1 | unpub. data |
| 12 Jun 2000 | Reserve Canal from near Lake Maurepas to near I-10 | 30.13155 | -90.54696 | 1 | unpub. data |
| mid-Jul 2000 | Near Point-Aux-Chene | 29.52771 | -91.43816 | 1 | unpub. data |
| 21–23 Jul 2000 | Bayou Lafourche from Raceland to Matthews | 29.63682 | -90.51146 | 1 | unpub. data |
| 12 Aug 2000 | edge of Gulf north of Pass A L'outre near mouth of Mississippi River | 29.20770 | -89.11896 | 1 | unpub. data |
| 4 Mar 2001 | Amite River | 30.27004 | -90.76221 | 1 | unpub. data |
| 4 Jun 2001 | Bogue Falaya near the old train trestle at the US 190 overpass in Covington | 30.48109 | -90.08441 | 1 | unpub. data |
| 9 Jul 2001 | Inner Harbor Canal Lock (Industrial Canal at Mississippi River) | 29.96034 | -90.02560 | 1 | unpub. data |
| 18 Aug 2001 | Calcasieu River, app. 1 mi east of I-210 bridge, Lake Charles | 30.21198 | -93.25891 | 1 | unpub. data |
| 15 Oct 2001 | Lake Boeuf | 29.87336 | -90.59723 | 1 | unpub. data |
| 22 Oct 2001 | Near Rigolets. St. Tammany Parish | 30.22490 | -89.79440 | 1 | unpub. data |
| Nov 2001 | Eden Isles, near entrance to Lake Pontchartrain | 30.21983 | -89.82734 | 1 | unpub. data |
| 2001 | near jetty connecting Lake Bourne with ship channel | 29.89557 | -89.75230 | 1 | unpub. data |
| 9 May 2002 | Bonnet Carre Spillway, southwest of Lake Pontchartrain | 30.04944 | -90.27500 | 1 | unpub. data |
| 11 May 2002 | Faciane Canal at 301 Carr Drive | 30.24177 | -89.84100 | 1 | unpub. data |
| 12 May 2002 | Irish Bayou Canal, town of Little Woods | 30.11610 | -89.86600 | 1 | unpub. data |
| 19 May 2002 | Faciane Canal at 411 Carr Drive | 30.23268 | -89.85360 | 1 | unpub. data |
| 1 Jun 2002 | Bay Eloi side of Mosquito Bayou | 29.79000 | -89.43000 | 1 | unpub. data |
| 9 Jun 2002 | Mandeville Harbor | 30.34100 | -90.09500 | 1 | unpub. data |
| 18 Jul 2002 | east side of sea wall (Mandeville) | 30.34100 | -90.09500 | 2 | unpub. data |

**APPENDIX I (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **LOUISIANA (continued)** | | | | |
| 20 Jul 2002 | between Tchefuncte River and Causeway near Mandeville | 30.38997 | -90.15536 | 1 | unpub. data |
| 23 Jul 2002 | Sunset Point (2 miles from harbor). Mandeville, Lake Pontchartrain | 30.34100 | -90.09500 | 2 | unpub. data |
| 23 Jul 2002 | marina at Causeway | 30.34100 | -90.09500 | 1 | unpub. data |
| 24 Jul 2002 | Bayou Castine, at Causeway | 30.34100 | -90.09000 | 1 | unpub. data |
| 24 Jul 2002 | Mandeville Harbor | 30.34100 | -90.09500 | 1 | unpub. data |
| 26 Jul 2002 | Sunset Point, Mandeville. Lake Pontchartrain | 30.34100 | -90.09500 | 1 | unpub. data |
| 29 Jul 2002 | Sunset Point, Mandeville, Lake Pontchartrain | 30.34100 | -90.09500 | 1 | unpub. data |
| 7 Aug 2002 | Sunset Point, Mandeville, Lake Pontchartrain | 30.34100 | -90.09500 | 2 | unpub. data |
| 12 Aug 2002 | Mandeville Harbor | 30.34100 | -90.09500 | 1 | unpub. data |
| 17 Aug 2002 | jetties at South Pass | 29.05165 | -89.20135 | 1 | unpub. data |
| 19 Aug 2002 | Sunset Point, Mandeville, Lake Pontchartrain | 30.34100 | -90.09500 | 2 | unpub. data |
| 19 Aug 2002 | Slidell | 30.22200 | -89.83000 | 1 | unpub. data |
| 28 Aug 2002 | Venice, in nearshore waters | 29.27643 | -89.35320 | 2(c/c) | unpub. data |
| Sep 2002 | mouth of Tchefuncte River at Madisonville | 30.37711 | -90.16016 | 1 | unpub. data |
| 9 Sep 2002 | Faciane Canal toward Paradise Island | 30.23472 | -89.85750 | 1 | unpub. data |
| 29 Sep 2002 | Green Point (near Fontainebleau State Park), Mandeville | 30.32368 | -90.03954 | 1 | unpub. data |
| 10 Oct 2002 | Mandeville Harbor | 30.32022 | -90.08827 | 4 | unpub. data |
| 8 Apr 2003 | Slidell | 30.24185 | -89.86494 | 1 | unpub. data |
| 1 Jun 2003 | Eden Isles | 30.21983 | -89.82777 | 1 | unpub. data |
| 2 Jun 2003 | Conway Canal, swimming towards Blind River | 30.22709 | -90.66263 | 1 | unpub. data |
| 6 Jun 2003 | Marina del Ray, Madisonville | 30.27028 | -89.77330 | 1 | unpub. data |
| 29 Jun 2003 | Natalbany River | 30.38901 | -90.50471 | 1 | unpub. data |
| 16 Jun 2003 | Goose Point (near Lacombe), St. Tammany Parish | 30.26179 | -89.93313 | 1 | unpub. data |
| 4 Jul 2003 | Bayou Paquet near its junction with Bayou Liberty, Slidell | 30.26330 | -89.85710 | 2(c/c?) | unpub. data |
| 13 Jul 2003 | in open water at southwestern tip of the Chandeleur Islands | 29.66000 | -88.90000 | 1 | unpub. data |
| 17 Jul 2003 | Mandeville Harbor | 30.34100 | -90.09500 | 1 | unpub. data |
| 30 Jul 2003 | Indian Landing Marina (Bayou Castine) | 30.35270 | -90.04800 | 2 | unpub. data |
| 16 Aug 2003 | Mouth of Cane Bayou, close to Lacombe | 30.32000 | -90.01000 | 1 | unpub. data |
| 22 Aug 2003 | Leisure Landing on Tickfaw River, end of Hwy 1037, close to Springfield | 30.36520 | -90.59480 | 1 | unpub. data |
| 25 Aug 2003 | North Shore Beach, outside Slidell and then in the North Shore canal | 30.21000 | -89.81000 | 2(c/c) | unpub. data |
| Sep 2003 | Mandeville Harbor | 30.34100 | -90.09500 | 1 | unpub. data |

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**APPENDIX I (continued)**

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **LOUISIANA (continued)** | | | | |
| Nov 2003 | Myrtle Grove, Plaquemines Parish | 29.64043 | -89.94495 | 1 | unpub. data |
| 6 May 2004 | northwest side of the Barataria Bay island, Plaquemines Parish | 29.37500 | -90.00000 | 1 | unpub. data |
| 10 May 2004 | Bayou Bienvenue, Mississippi outlet | 29.27387 | -89.15762 | 1 | unpub. data |
| 21 May 2004 | Gulf outlet of Mississippi River | 29.85053 | -89.65748 | 1 | unpub. data |
| 1 Jul 2004 | Amite River, app. 2 mi north of Port Vincent | 30.34529 | -90.87423 | 1 | unpub. data |
| 11 Aug 2004 | Amite River | 30.34178 | -90.86118 | 1 | unpub. data |
| | **TEXAS** | | | | |
| Summer 1912 | Laguna Madre | 27.72300 | -97.31702 | N/A | Gunter (1941, 1954), Powell and Rathbun (1984) |
| 1913 or 1914 | 2 mi above mouth of Rio Grande River | 25.98987 | -97.12387 | 2 | Gunter (1941), Powell and Rathbun (1984) |
| 1915-1919 | Mouth of Rio Grande River | 25.96531 | -97.12668 | N/A | Gunter (1941) |
| 2 Oct 1976[3] | Fish Pass near Corpus Christi, Mustang Island | 27.64888 | -97.24066 | 1 | Powell and Rathbun (1984), Fernandez and Jones (1990) |
| late May 1977 | North Padre Island, off 2nd sandbar, in surf southeast of Padre Island Beach Hotel | 26.52291 | -97.26163 | 1 | unpub. data |
| late Jul 1977 | North Padre Island, off 2nd sandbar, in surf southeast of Padre Island Beach Hotel | 26.48997 | -97.24366 | 1 | unpub. data |
| 9 Nov 1992 | Port Isabel, lower Laguna Madre | 26.07611 | -97.22222 | 1 | unpub. data |
| 7 Jul 1994 | lower Laguna Madre, near swing bridge, South Padre Island | 26.07083 | -97.20000 | 1 | unpub. data |
| Summer 1994 | Arroyo Colorado | 28.61667 | -95.95000 | 1 | unpub. data |
| Sep 1994 | Laguna Madre | 27.71177 | -97.28558 | 1 | unpub. data |
| Aug 1995 | Port Mansfield Pass | 26.55000 | -97.30000 | 1 | unpub. data |
| 3–8 Sep 1995 | west Galveston Bay near San Luis Pass | 29.08333 | -95.13333 | 2(c/c) | Schiro and Fertl (1995) |
| 15 Sep 1995 | Galveston, 8 km east of San Luis Pass | 29.13333 | -95.08333 | 2 c/c) | Schiro and Fertl (1995) |
| 18 Sep 1995 | Sea Isle, Galveston | 29.13333 | -95.08333 | 2 c/c) | Schiro and Fertl (1995) |
| 18 Sep 1995 | North Deer Island, Galveston Bay | 29.28333 | -94.93333 | 2(c/c) | Schiro and Fertl (1995) |
| early Oct 1995 | Port Mansfield area | 26.55000 | -97.38333 | 2(c/c) | Schiro and Fertl (1995) |
| 8 Oct 1995 | Laguna Madre, Gulf Intracoastal Waterway near marker 5 | 27.70129 | -97.27135 | 1 | unpub. data |
| 15 Oct 1995 | 35 mi south of John F. Kennedy Causeway, Laguna Madre | 27.69538 | -97.31998 | 1 | Schiro and Fertl (1995) |

[3]Powell and Rathbun (1984) listed an incorrect date; Sirenia Project Files have date as 2 Oct 1976

| Date | Location description | Latitude | Longitude | No. Individ. | Source |
|---|---|---|---|---|---|
| | **TEXAS (continued)** | | | | |
| 18 Oct 1995 | near Paul's Mott Lake behind San Jose Island, Aransas Bay | 27.96667 | -96.98333 | 1 | unpub. data |
| 25 Oct 1995 | Barney Davis Power Plant, Corpus Christi | 27.60250 | -97.30500 | 1 | unpub. data |
| 26 Oct 1995 | fishing pier at Naval Air Station Corpus Christi | 27.71018 | -97.30112 | 1 | unpub. data |
| 31 Oct 1995 | Naval Station Ingleside | 27.81833 | -97.20833 | 1 | unpub. data |
| 2 Nov 1995 | Rockport Harbor and boat basin | 28.02952 | -97.02458 | 1 | unpub. data |
| 6 Nov 1995 | Koch Refinery on the La Quinta Channel, Corpus Christi | 27.81432 | -97.39074 | 1 | unpub. data |
| 8 Nov 1995 | near Texas State Aquarium | 27.83000 | -97.39000 | 1 | unpub. data |
| 11 Nov 1995 | south jetty of the Aransas Pass | 27.83667 | -97.04500 | 1 | unpub. data |
| 12 Nov 1995 | University of Texas Marine Science boat basin | 27.83667 | -97.05167 | 1 | unpub. data |
| 12 Nov 1995 | Port Aransas Municipal Harbor | 27.83833 | -97.06667 | 1 | unpub. data |
| 27 Nov 1995 | 69th Street Wastewater Treatment Plant, Houston | 29.70000 | -95.28333 | 1 | Schiro and Fertl (1995) |
| 6 Dec 1995 | 69th Street Wastewater Treatment Plant, Houston | 29.70000 | -95.28333 | 1 | unpub. data |
| May 1996 | San Leon, Galveston Bay | 29.50000 | -94.91667 | 1 | unpub. data |
| 14 Jul 2001 | Rockport | 28.04100 | -97.02261 | 1 | unpub. data |
| late Jul 2001 | University of Texas Marine Science Institute boat basin | 27.83667 | -97.05167 | 1 | unpub. data |
| 11 Sep 2001 | Hampton's Landing boat basin in Aransas Pass | 27.88034 | -97.13994 | 1 | unpub. data |
| 23 Sep 2001 | inlet between Texas State Aquarium and Lexington Museum | 27.81833 | -97.38472 | 1 | Beaver (2001) |
| 3 Oct 2001 | near Texas State Aquarium | 27.83000 | -97.39000 | 1 | unpub. data |
| 11 Oct 2001 | near Portland | 27.83569 | -97.35056 | 1 | unpub. data |
| 26 Oct 2001 | near Portland | 27.85083 | -97.35028 | 1 | unpub. data |
| 14 Nov 2001 | dock at Valero Refining Company. Corpus Christi | 27.81833 | -97.47500 | 1 | unpub. data |
| 29 Nov 2001 | near Portland | 27.82246 | -97.43557 | 1 | unpub. data |
| 30 Nov 2001 | near Portland | 27.80821 | -97.38879 | 1 | unpub. data |
| 5 Dec 2001 | near Portland | 27.84852 | -97.32349 | 1 | unpub. data |
| 12 Dec 2001 | near Portland | 27.83569 | -97.35056 | 1 | unpub. data |
| 8 Oct 2003 | Port Isabel, lower Laguna Madre | 27.69148 | -97.25848 | 1 | unpub. data |
| 27 Jun 2004 | Port Aransas Municipal Harbor/Trout Street Marina | 27.84000 | -97.06000 | 1 | unpub. data |
| 20 Jul 2004 | south jetty of the Aransas Pass | 27.83785 | -97.05050 | 1 | unpub. data |
| 21 Jul 2004 | south jetty of the Aransas Pass | 27.83417 | -97.04583 | 1 | unpub. data |
| 26 Jul 2004 | Padre Isles residental channel off Fortuna Bay Drive | 27.59117 | -97.24333 | 1 | unpub. data |
| 2 Aug 2004 | near Hampton's Landing, Aransas Pass | 27.88534 | -97.13994 | 1 | unpub. data |
| 6 Aug 2004 | south jetty of the Aransas Pass | 27.83333 | -97.04167 | 1 | unpub. data |
| 7 Aug 2004 | near Reynold's Aluminum Plant (La Quinta Channel) | 27.83317 | -97.26680 | 1 | unpub. data |
| 15 Aug 2004 | Willacy County Navigation District Ramp (south ramp) | 26.5854 | -97.4471 | 1 | unpub. data |

# APPENDIX II

Mortality records for manatees in the northern Gulf of Mexico, west of Florida. Each record represents a single individual. FWC MMPL - Florida Fish and Wildlife Commission, Marine Mammal Pathobiology Lab, St. Petersburg, Florida; FWSJX—US Fish and Wildlife Service, Jacksonville, Florida; LA—Louisiana; PI—Padre Island, Texas; PO—Port O'Connor, Texas; SHCM—Spring Hill College, Mobile, Alabama; TCWC—Texas Wildlife Cooperative Collection, Texas A&M University, College Station, Texas; USNM—US National Museum, Washington, DC; VM—belongs to Houston Museum of Natural Science, Vertebrate Mammalogy department, Houston, Texas.

| Date | Location description | Latitude | Longitude | Specimen number | Cause of death | Source | Notes |
|---|---|---|---|---|---|---|---|
| **ALABAMA** | | | | | | | |
| 13 Jan 1992 | Mobile Bay, east junction of Highway 98 and County Road 1 | 30.41660 | −87.90083 | SHCM 119 | cold stress | unpub. data | 3.2 m male; 554.5 kg |
| 20 Jul 2000 | near the Beer Cans Rig off Mobile | 30.16670 | −88.03330 | N/A | N/A | unpub. data | floater |
| 26 Feb 2002 | south shore of Dog River | 30.58170 | −88.11030 | SHCM 350 | cold stress | unpub. data | 2.6 m male |
| 4 Jan 2004 | Rabbit Creek near end of Le Blanc Road | 30.58080 | −88.12972 | SHCM 370 | N/A | unpub. data | 3.3 m male |
| **MISSISSIPPI** | | | | | | | |
| 3 Jan 1980 | 1/4 mile east of eastern end Ship Island, north shore | 30.20000 | −88.86667 | Sirenia Project salvage records M-179 | starvation and cold stress | Gunter and Corcoran (1981), Powell and Rathbun (1984) | 2.3 m female; 240.4 kg |
| 8 Feb 2001 | Horn Island, south beach bout 1/2 mi east of ranger station trail | 30.22890 | −88.65440 | FWC MMPL MS2001UNK | cold stress | unpub. data | 2.8 m individ. |
| 17 Feb 2002 | shore of Back Bay in Biloxi (6500 Old Bay Rd) | 30.41580 | −88.99417 | FWC MMPL MS2002UNK | unknown | unpub. data | 2.4 m male; 680 kg |
| **LOUISIANA** | | | | | | | |
| Jan 1929 | McFaddens Beach, Calcasieu Lake | 29.88880 | −93.28894 | USNM 257406 | dynamite blastings by oil survey party | Gunter (1941, 1954), Lowery (1943), Moore (1951a), Lowery (1974), Gunter and Corcoran (1981), Hartman (1974), Powell and Rathbun (1984) | skull length of 381 mm |
| 15 Jun 1995 | canal between middle Bayou (off North Pass) and Tangipahoa River | 30.33860 | −90.31247 | N/A | human interaction | unpub. data | 2.7 m individ. |
| *22 Jul 1995 | App. 3 mi southeast of Breton Sound Marina in canal, near Hopedale, St. Bernard Parish | 29.82270 | −89.61985 | LA 9501 | human interaction | unpub. data | 2.7 m individ. > 454 kg |

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**APPENDIX II (continued)**

| Date | Location description | Latitude | Longitude | Specimen number | Cause of death | Source | Notes |
|---|---|---|---|---|---|---|---|
| **LOUISIANA (continued)** | | | | | | | |
| 5 Dec 1995 | Vermilion Bay | 29.75078 | –92.15983 | N/A | N/A | unpub. data | floater |
| 31 Jan 1996 | Bally's Casino Lakeshore Resort, southern shore of Lake Pontchartrain | 30.03420 | –90.00216 | FWSJX 9601 | caught in pumping station pipe | unpub. data | 3.2 m individ. |
| 9 Sep 1997 | Locust Bayou near Pt. Au Fer, near the Mobil Oil facility, Terrebonne Parrish | 29.30050 | –91.29127 | LA-035-97 | undetermined | unpub. data | 3.2 m male |
| 14 Dec 1999 | Bayou Patout, northeast of Weeks Island, west of junction of Patout Canal and Iberia/St. Mary Canal | 29.84040 | –91.74000 | N/A | N/A | unpub. data | > 3.0 m male |
| 11 Dec 2000 | Creole Bayou area between Atchafalaya and 4 League Bays | 29.40890 | –91.20928 | N/A | N/A | unpub. data | 1.8–2.7 m individ.; 227–454 kg |
| 10 Dec 2001 | Lake Borgne, 17 mi southeast of Bay St. Louis | 30.52530 | –89.39388 | N/A | N/A | unpub. data | N/A |
| 21 Feb 2001 | Intracoastal Waterway, 7–8 mi west of Larose | 29.56040 | –90.49310 | N/A | N/A | unpub. data | N/A |
| early Apr 2002 | Pointe a la Hache (Plaquemines Parish) | 29.55230 | –89.88416 | N/A | N/A | unpub. data | N/A |
| 29 May 2002 | Eloi Bay, near Shell Beach | 29.73760 | –89.46840 | N/A | N/A | unpub. data | carcass disappeared |
| 26 Mar 2003 | Raceland, in canals north of Lake Field | 29.66030 | –90.59994 | N/A | N/A | unpub. data | N/A |
| 3 Aug 2003 | mouth of Tchefuncte River | 30.37710 | –90.16016 | N/A | N/A | unpub. data | > 454 kg |
| end of Feb 2004 | Biloxi Wildlife Management Area | 30.04292 | –89.53004 | N/A | N/A | unpub. data | N/A |
| **TEXAS** | | | | | | | |
| Jul 1928 | north end of Shellbank Reef in Copano Bay near town of Bayside | 28.83330 | –95.60000 | TCWC 1528 | N/A | Gunter (1941, 1944), Moore (1951a), Hartman (1974), Powell and Rathbun (1984), Fernandez and Jones (1990), Jefferson and Baumgardner (1997) | 578 cm male |
| prior to 1929 | San Jose Island, east of Rockport | 27.99175 | –97.04487 | VM.132 and VM.133 | N/A | Baumgardner and Brooks (2001) | N/A |
| 1937 | mouth of Cow Bayou, near Sabine Lake, Cameron Parish | 29.86300 | –93.80562 | N/A | N/A | Gunter (1941, 1944), Lowery (1943[1],1974), Hartman (1981), Powell and Rathbun (1984) | male |

[1]Lists incorrect date of July 1928 for this account.

| Date | Location description | Latitude | Longitude | Specimen number | Cause of death | Source | Notes |
|---|---|---|---|---|---|---|---|
| **TEXAS (continued)** | | | | | | | |
| 4 Feb 1986 | 1 mi west of Caplen. Bolivar Peninsula | 29.48560 | –94.53527 | TCWC 49000 | 7 holes and a recent rope mark | O'Shea (1988)[2]; Fernandez and Jones (1990), Jefferson and Baumgardner (1997)[2] | 2.7 m male |
| 30 Nov 1992 | 40 km north of Brazos Santiago Pass | 26.43670 | –97.22833 | PI53 | N/A | unpub. data | > 2.7 m individ.; 544 kg |
| 31 Oct 1993 | 15.7 mi east of beach turnoff (Matagorda County Jetty Park) | 28.71670 | –95.70600 | PO274 | Human interaction (bullet hole in head caused death; old bullet in rib; caught in monofilament) | unpub. data | 2.5 m male; 172.2 kg |
| 18 Dec 1999 | 3 mi north of Wynn access road, Calhoun County | 28.14500 | –96.75350 | PO375 | starvation and cold stress | unpub. data | 2.6 m female |
| 14 Oct 2001 | Gulf Intracoastal Waterway by County Park, Sargent Beach | 28.76400 | –95.62917 | PO386 | not determinable (malnourished?) | Beaver (2001) | 3.0 m male; > 454 kg |

[2]Source provides incorrect date of 1983.

# Gulf and Caribbean Research

High Cyanobacterial Abundance in Three Northeastern Gulf of Mexico Estuaries

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Recommended Citation

# HIGH CYANOBACTERIAL ABUNDANCE IN THREE NORTHEASTERN GULF OF MEXICO ESTUARIES

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*ABSTRACT* Aquatic phytoplankton comprise a wide variety of taxa spanning more than 2 orders of magnitude in size, yet studies of estuarine phytoplankton often overlook the picoplankton, particularly chroococcoid cyanobacteria (cf. *Synechococcus*). Three Gulf of Mexico estuaries (Apalachicola Bay, FL; Pensacola Bay, FL; Weeks Bay, AL) were sampled during summer and fall 2001 to quantify cyanobacterial abundance, to examine how cyanobacterial abundance varied with hydrographic and nutrient distributions, and to estimate the contribution of cyanobacteria to the bulk phytoplankton community. Cyanobacterial abundances in all 3 estuaries were high, averaging $0.59 \pm 0.76 \times 10^9$ L$^{-1}$ in Apalachicola Bay, $1.7 \pm 1.2 \times 10^9$ L$^{-1}$ in Pensacola Bay and $2.4 \pm 1.9 \times 10^9$ L$^{-1}$ in Weeks Bay (mean ± standard deviation). Peak abundances typically occurred in the oligohaline zone (low salinity estuarine zone) during the summer. Freshwater sites had nearly undetectable abundances, and marine sites had abundances several-fold lower than the oligohaline zone. When converted to equivalent chlorophyll *a* concentrations, cyanobacteria comprised a large fraction of the total phytoplankton biomass, at times approaching 100% in all 3 systems. These observations clearly indicate a cyanobacterial community of estuarine origin that can make up a large proportion of phytoplankton biomass.

## INTRODUCTION

Phytoplankton are responsible for about 40% of global primary production and form the base of the aquatic food web; they are thus critically important mediators of carbon and energy (Falkowski 1994). Quantitative measures of phytoplankton biomass, size distribution, and community composition are important indicators of the trophic state of aquatic systems and provide insight into the environmental forcings that affect phytoplankton dynamics (Chisholm, 1992). Phytoplankton taxonomic and size composition can also provide insight into the trophic transfer to zooplankton grazers and help predict the resulting zooplankton community composition (Hansen et al. 1994).

In the open ocean, phytoplankton biomass and production are typically dominated by the picophytoplankton (phytoplankton < 2 μm), which are largely comprised of cyanobacteria (e.g., *Synechococcus*) and prochlorophytes (Li 1998). In estuaries, however, the importance of picophytoplankton is not well understood, because estuarine studies often overlook cyanobacteria. A commonly used method for enumerating phytoplankton relies on settling of organisms from a water sample (Utermol 1958). However, particles of 1–2 μm are effectively colloidal and do not sink. Therefore, such studies are biased towards organisms larger than 5–10 μm, thereby overlooking the potential contribution of picophytoplankton (e.g., Livingston 2001, 2003). Nevertheless, there is a growing body of literature showing that estuaries have high cyanobacterial abundances, particularly during the summer, but often their contribution to the total phytoplankton biomass is relatively small (Pinckney et al. 1998, Ning et al. 2000). Notable exceptions include studies in subtropical systems such as Florida Bay (Phlips et al. 1999) and Pensacola Bay (Murrell and Lores 2004), where cyanobacteria can dominate the phytoplankton biomass.

The purpose of this study was to enumerate cyanobacteria in 3 Gulf of Mexico (GOM) estuaries: Apalachicola Bay, Florida; Pensacola Bay, Florida; and Weeks Bay, Alabama. We examined their distribution along the salinity gradient and examined their relationship with chlorophyll *a* (Chl *a*) and dissolved inorganic nitrogen (DIN) concentrations. Additionally, we estimated the cyanobacterial contribution to total Chl *a*, using an estimate of their cell-specific Chl *a* content. Data on cyanobacterial abundances and Chl *a* from Pensacola Bay are a subset of a larger dataset originally reported in Murrell and Lores (2004) and were included here for comparative purposes.

## MATERIALS AND METHODS

### Study sites

The 3 estuaries chosen for this study are all located along the northeastern coastline of the GOM (Figure 1) and therefore share similar patterns of solar radiation and rainfall. All sites are quite shallow, averaging from 2 to 3 m depth, but vary in estuarine area, watershed area and freshwater flow (Table 1). Apalachicola Bay, located in the middle of the Florida panhandle, is 593 km² in size and receives freshwater from the Apalachicola River. Land cover in the Apalachicola portion of the watershed is pri-
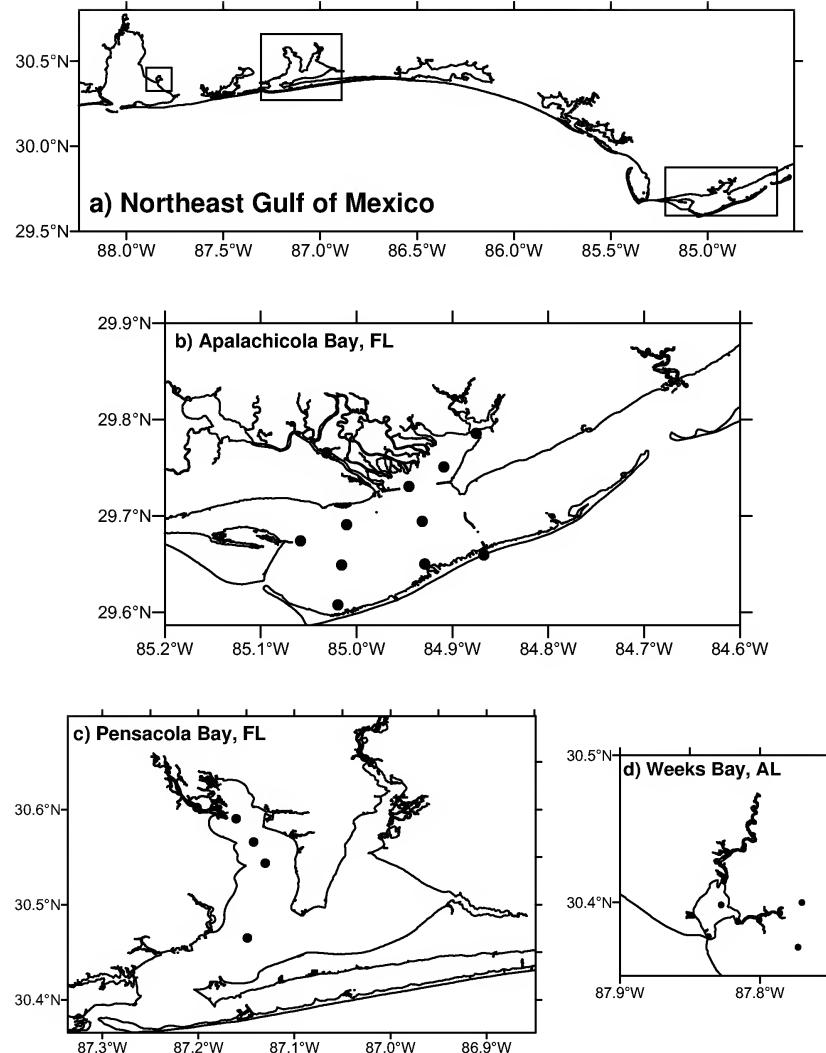
**Figure 1. a) Map of study area in the northeastern Gulf of Mexico. Inset maps are included for b) Apalachicola Bay, FL; c) Pensacola Bay, FL; and d) Weeks Bay, AL.**

marily forest, including pine flatwoods and bottomland hardwood, with little residential and commercial development. Pensacola Bay is a moderately sized (370 km²) estuary in the western panhandle of Florida. The major freshwater source is the Escambia River (180 m³ s⁻¹), which empties into the western side of the system. Other rivers include the Yellow (45 m³ s⁻¹), and Blackwater (9.2 m³ s⁻¹), which flow into the eastern side of the system. The watershed is comprised of pine forests (74%), croplands (12%), pastures (7%) and urban development (2%) that is concentrated near the shoreline of the bay. Weeks Bay is a sub-estuary of Mobile Bay, Alabama, and is much smaller (7 km²) than Pensacola and Apalachicola Bays. The Fish River contributes 90% of the freshwater flow into Weeks Bay, and at the seaward end, Weeks Bay empties into

Mobile Bay. Land use is dominated by agriculture, both timber production and cropland, which together represent 68% of the land use in Baldwin County, where Weeks Bay is located (Arcenaux 1996). Agricultural lands are rapidly being converted to suburban developments as population growth increases throughout the county.

**Field collection**

Samples were collected during summer and fall 2001 (Table 2). In general, sampling sites were oriented along major salinity gradients. Apalachicola Bay was sampled on 3 dates (Sep, Oct, Nov) at seven sites. In November, 4 additional sites were sampled. Pensacola Bay was sampled on five dates (Jul, Aug, Sep, Oct, Nov) at 5 sites on the western side of the system. The Pensacola Bay data are a

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**TABLE 1**

Summary of key physical and environmental characteristics of the 3 Gulf of Mexico estuaries sampled in this study. Rainfall and river flow data are long term means. Residence times are calculated via the fraction of freshwater method of Dyer 1973.

| Estuary | Mean annual rainfall (cm) | Estuarine area (km$^2$) | Watershed area (km$^2$) | Watershed area: Estuarine area | Mean depth (m) | Mean river flow (m$^3$ s$^{-1}$) | Mean residence time (d) |
|---|---|---|---|---|---|---|---|
| Apalachicola Bay | 143 | 593 | 51000 | 86 | 2.9 | 710 | 6 |
| Pensacola Bay | 163 | 370 | 13500 | 37 | 3.3 | 234 | 25 |
| Weeks Bay | 165 | 7.0 | 510 | 73 | 2.0 | 3.4 | 6 |

**TABLE 2**

Station names and locations sampled in this study. The mean salinity from all sampling dates and stations is provided to indicate the station's relative position within the estuary.

| Estuary | Station | Latitude | Longitude | Mean salinity (psu) |
|---|---|---|---|---|
| Apalachicola Bay | Apalachicola River | 29° 45.93'N | 85° 01.87'W | 2 |
| | ANERR 5 | 29° 41.48'N | 85° 00.63'W | 18 |
| | Dry Bar | 29° 40.48'N | 85° 03.50'W | 22 |
| | ANERR 4 | 29° 38.96'N | 85° 00.93'W | 23 |
| | ANEER 8 | 29° 43.85'N | 84° 56.71'W | 23 |
| | East Bay | 29° 47.15'N | 84° 52.52'W | 24 |
| | ANERR 3 | 29° 36.47'N | 85° 01.17'W | 25 |
| | ANERR 9 | 29° 45.08'N | 84° 54.52'W | 27 |
| | ANERR 6 | 29° 39.02'N | 84° 55.73'W | 30 |
| | ANERR 7 | 29° 41.67'N | 84° 55.89'W | 30 |
| | Gulf | 29° 39.58'N | 84° 52.03'W | 31 |
| Pensacola Bay | P01 | 30° 33.13'N | 87° 12.09'W | 1 |
| | P02 | 30° 32.42'N | 87° 09.64'W | 10 |
| | P03 | 30° 30.95'N | 87° 08.56'W | 15 |
| | P04 | 30° 29.62'N | 87° 07.83'W | 17 |
| | P06 | 30° 24.91'N | 87° 08.94'W | 21 |
| Weeks Bay | Weeks Creek, Upper | 30° 22.17'N | 87°46.37'W | 0 |
| | Magnolia River, Upper | 30° 23.99'N | 87° 46.20'W | 1 |
| | Fish River | 30° 26.18'N | 87° 48.71'W | 3 |
| | Waterhole Branch | 30° 26.04'N | 87° 49.39'W | 3 |
| | Turkey Branch | 30° 25.67'N | 87° 49.84'W | 4 |
| | Lulu Dock | 30° 24.88'N | 87° 49.55'W | 5 |
| | Weeks Creek, Lower | 30° 23.56'N | 87° 47.15'W | 6 |
| | Nolte Creek | 30° 23.29'N | 87° 48.03'W | 8 |
| | Magnolia River, Lower | 30° 23.21'N | 87° 48.95'W | 9 |
| | Mid Bay | 30° 23.90'N | 87° 49.65'W | 11 |
| | Mouth | 30° 22.60'N | 87° 50.20'W | 14 |

subset of a 2 year study examining phytoplankton and zoo-plankton dynamics previously reported in Murrell and Lores (2004). The dates chosen for inclusion in this study overlap the time frame of the other 2 sites. Weeks Bay was sampled on 3 dates (Jul, Sep, Nov) at 11 sites.

Water samples were collected from the surface layer (top 0.5 m) into clean polyethylene bottles and processed at the lab within hours. Salinity (psu) was measured either with a Hyrolab multimeter (Pensacola Bay) or with a Thermo Orion Model 150A+ conductivity meter (Apalachicola and Weeks Bays). Chl $a$ samples were filtered onto Whatman 25 mm GF/F filters (50 to 200 ml) and frozen ($-20$ °C) until analysis. Chl $a$ was extracted in 90% acetone (Strickland and Parsons 1972), and fluorescence was measured with a Turner Designs TD 700 fluorometer calibrated using commercially available Chl $a$ standards (Sigma Chemicals). Cyanobacterial samples were collected into 20 ml vials, fixed with 2% final concentration formaldehyde and stored at 4 °C until cell counts were performed via epifluorescence microscopy, as described in Murrell and Lores (2004). Samples for nutrients ($NH_4^+$, $NO_2^-$, $NO_3^-$, $PO_4^{3-}$) were stored in clean glass or HDPE vials and analyzed using standard methods (APHA 1989). DIP (dissolved inorganic phosphorus) is used to denote $PO_4^{3-}$, while DIN is the sum of $NO_2^- + NO_3^- + NH_4^+$.

## RESULTS

Weather conditions during this study were typical for summer and early fall in the region, including warm water temperatures (28–30 °C) and episodic rainfall events due to thunderstorm activity. River flow during this period was lower than normal for the region. Mean flows (from July through November 2001) were 60% (Apalachicola River), 72% (Fish River), and 89% (Escambia River) of long term means for the same time window (http://water.usgs.gov).

Over all sites and dates, Chl $a$ concentration varied widely from 1 to > 250 µg $L^{-1}$ (Table 3). Weeks Bay had the highest Chl $a$ concentration peaking at over 200 µg $L^{-1}$ at the Turkey Branch site, but also exceeding 100 µg $L^{-1}$ at several other sites (Figure 2). In contrast, Chl $a$ in Apalachicola Bay and Pensacola Bay never exceeded 20 µg $L^{-1}$ and had ranges and means similar to each other (Figure 2). One common finding in all 3 systems was that Chl $a$ tended to peak at the mid-estuarine sites on a given date (Figure 2). DIN concentrations ranged from below detection to 148 µM, exhibiting a typical spatial pattern with highest concentrations at the freshwater sites decreasing along the freshwater to marine estuarine gradient (Figure 3). Weeks Bay had by far the highest DIN concen-

trations, with peak concentrations at the Upper Magnolia River site, ranging 94.2 to 148 µM (Figure 3). As with Chl $a$, Apalachicola and Pensacola Bays had similar but much lower DIN concentrations, rarely exceeding 20 µM. DIP concentrations were generally low in all estuaries, never exceeding 1 µM (Table 3), and there were no obvious DIP-salinity gradients (data not shown). Cyanobacterial abundance varied by over 3 orders of magnitude from 0.004 to 5.8 X $10^9$ $L^{-1}$ and, similar to bulk Chl $a$, were generally most abundant at the mid-estuarine sites (Figure 4), peaking at salinities near 5–10 psu in Weeks Bay, 10 psu in Pensacola Bay, and 22 psu in Apalachicola Bay (Figure 5a). Similar to DIN and Chl $a$ concentrations, mean cyanobacterial abundance was highest in Weeks Bay and lower in Apalachicola and Pensacola Bays (Table 3, Figure 4). However, in contrast with DIN, the freshwater sites had the lowest cyanobacterial abundances, usually one or 2 orders of magnitude lower than nearby estuarine sites. This pattern was most evident in Pensacola Bay (P01) and Weeks Bay (Weeks Creek, Magnolia River). At the marine sites, cyanobacteria abundances were lower than at the mid-estuarine sites, but not nearly as low as the freshwater sites. In Apalachicola Bay, only the East Bay site had high cyanobacterial abundances, averaging 2.3 X $10^9$ $L^{-1}$, 2 to 3 times higher than the other sites. In contrast, Weeks Bay and Pensacola Bay had high cyanobacterial abundances at most estuarine sites, peaking at 5.8 X $10^9$ $L^{-1}$ and 4.6 X $10^9$ $L^{-1}$, respectively.

Although there were only 3 sampling dates, there was a consistent temporal pattern in Weeks Bay and Pensacola Bay (Figure 4). In general, cyanobacterial abundance peaked during summer when temperatures are warmest (ca. 30 °C). In Pensacola Bay, peak abundances occurred during August, whereas, in Weeks Bay, a similar peak occurred during July (there was no August sampling in Weeks Bay). This temporal pattern was not evident in Apalachicola Bay where cyanobacterial abundances were similar on all dates; however, this may be due to inadequate sampling earlier in the summer, as the first sampling date was not until September.

In order to gauge the importance of the cyanobacterial component of the phytoplankton community, we converted cyanobacterial abundance to equivalent Chl $a$ concentration using a factor of 3.4 fg chl $a$ cell$^{-1}$ (see Murrell and Lores 2004). Cyanobacterial Chl $a$ was then normalized to the total Chl $a$ concentration and plotted as a function of salinity (Figure 5b). This analysis showed that cyanobacteria contributed a large fraction of the total Chl $a$, especially in the low- to mid-salinity zone of the all 3 estuaries. In Weeks Bay, for example, many values were at or near 100%, suggesting that virtually all of the phyto-

**TABLE 3**

Mean values and ranges for salinity, DIN, DIP, Chl *a*, and cyanobacterial abundances during 2001.

| Estuary | Date | # Sites | Salinity (psu) | | DIN (µM) | | DIP (µM) | | Chl *a* (µg L$^{-1}$) | | Cyanobacteria (X 10$^9$ L$^{-1}$) | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Mean | Range | Mean | Range | Mean | Range | Mean | Range | Avg. | Range |
| Apalachicola Bay | 14 Sep | 7 | 25.1 | 2.5–32.0 | 1.7 | 0.0–10.3 | 0.39 | 0.09–0.58 | 9.9 | 4.9–17.7 | 0.77 | 0.11–3.10 |
| | 12 Oct | 7 | 26.4 | 2.4–33.1 | 3.0 | 0.0–20.5 | 0.26 | 0.10–0.52 | 6.4 | 2.3–12.7 | 0.46 | 0.14–1.42 |
| | 20 Nov | 11 | 21.1 | 1.8–28.5 | 3.0 | 0.0–21.3 | 0.32 | 0.22–0.43 | 3.3 | 1.6–6.2 | 0.48 | 0.11–2.35 |
| | Avg. | | 24.2 | | 2.6 | | 0.32 | | 6.4 | | 0.57 | |
| Pensacola Bay | 10 Jul | 5 | 8.3 | 0.0–17.9 | 5.2 | 0.7–10.8 | 0.07 | 0.01–0.19 | 12.0 | 4.2–18.2 | 1.50 | 0.07–2.92 |
| | 8 Aug | 5 | 8.2 | 0.4–19.8 | 4.2 | 0.2–8.3 | 0.07 | 0.06–0.09 | 11.2 | 4.8–15.7 | 2.41 | 0.12–4.58 |
| | 11 Sep | 5 | 7.9 | 0.1–13.4 | 3.9 | 0.3–8.6 | 0.07 | 0.01–0.17 | 10.4 | 1.9–16.5 | 1.68 | 0.02–2.79 |
| | 16 Oct | 5 | 17.6 | 0.5–25.7 | 6.8 | 1.0–15.9 | 0.06 | 0.01–0.15 | 9.2 | 1.5–15.6 | 0.46 | 0.01–0.73 |
| | 13 Nov | 5 | 22.2 | 4.9–29.2 | 6.6 | 2.7–18.8 | 0.08 | 0.01–0.13 | 5.0 | 1.9–8.5 | 0.67 | 0.04–1.15 |
| | Avg. | | 12.9 | | 5.3 | | 0.07 | | 10.4 | | 1.35 | |
| Weeks Bay | 19 Jul | 11 | 4.0 | 0.1–11.0 | 17.1 | 0.0–148.1 | 0.25 | 0.04–0.45 | 25.2 | 1.0–99.5 | 3.04 | 0.06–5.79 |
| | 4 Sep | 11 | 3.5 | 0.0–7.8 | 23.1 | 0.0–94.2 | 0.47 | 0.17–0.90 | 75.1 | 2.9–253.6 | 1.64 | 0.004–5.20 |
| | 27 Nov | 11 | 9.6 | 0.1–22.0 | 46.6 | 1.4–139.3 | 0.20 | 0.01–0.90 | 24.6 | 0.9–171.0 | 1.23 | 0.02–2.84 |
| | Avg. | | 5.7 | | 28.9 | | 0.31 | | 25.2 | | 1.97 | |
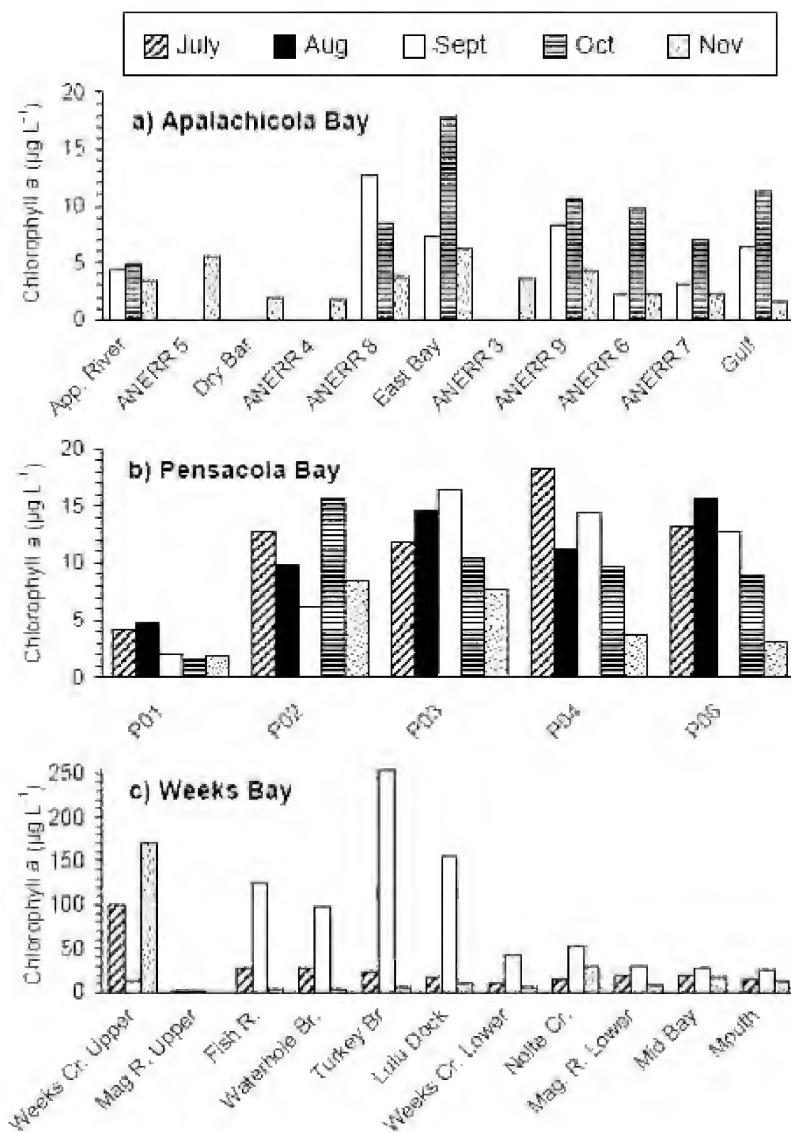
Figure. 2. Chl *a* concentration at all sites and dates sampled during the study: a) Apalachicola Bay, FL; b) Pensacola Bay, FL; c) Weeks Bay, AL (note different scaling). The sites were arranged in order of increasing mean salinity for each system as listed in Table 2.

plankton was comprised of cyanobacteria. In the other 2 systems, this fraction was not usually so high, but frequently exceeded 50%. Interestingly, when averaged across all sites and dates, the percentage of cyanobacterial Chl *a* was similar in all 3 systems; 31% for Apalachicola Bay, 39% for Pensacola Bay, and 36% for Weeks Bay. However, these global means can be considered biased low because the freshwater sites are clearly unsuitable habitat for cyanobacteria, where they contribute virtually 0% to total Chl *a*. Including only estuarine sites (mean salinity > 2, Table 2), the mean cyanobacterial contribution to total Chl *a* increased to 32%, 47% and 43%, respectively.

## DISCUSSION

The physical settings of the 3 estuarine systems (Table 1) have important similarities (e.g., rainfall, water depth) and differences (e.g., estuarine area, watershed area, freshwater flow) which help provide a context for interpreting the biological and chemical data. Apalachicola Bay is the largest system, with the largest watershed and is least impacted by anthropogenic nutrient inputs, as indicated by the low mean DIN at the freshwater source (mean 17.4 μM). Baywide mean Chl *a* and cyanobacterial concentrations were lower than those of the other 2 estuaries. Apalachicola Bay has a strong marine influence and a rel-
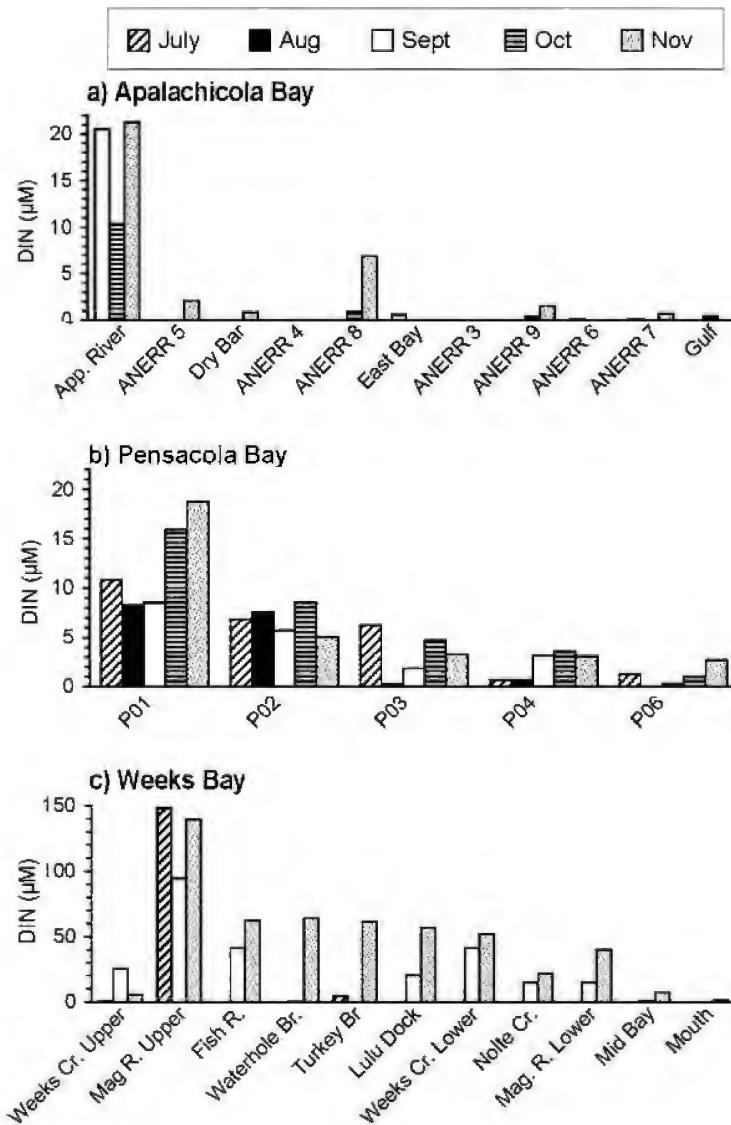
Figure 3. Dissolved inorganic nitrogen (DIN) concentration at all sites and dates sampled during the study: a) Apalachicola Bay, FL; b) Pensacola Bay, FL; c) Weeks Bay, AL (note different scaling). The sites were arranged in order of increasing mean salinity for each system as listed in Table 2.

atively short residence time (6 d), explaining the high mean salinity (24.2). The rapid Gulf exchange probably acts to dilute nutrient, Chl *a*, and cyanobacterial concentrations.

Pensacola Bay is intermediate in size, with moderate anthropogenic impacts from the watershed. Exchange with the GOM is narrowly constricted at Pensacola Pass, contributing to its relatively long residence time (25 d) and a lower mean salinity (12.9) than Apalachicola Bay. At the Escambia River site (P01) DIN averaged 12.5 µM, somewhat lower than the Apalachicola River mean; however, non-riverine sources of DIN (e.g. sewage treatment plants, urban storm-water runoff) are relatively more important in

Pensacola Bay, given the relatively high human population (ca. 300,000 people) surrounding the bay. This may in part explain the higher bay-wide mean DIN, Chl *a*, and cyanobacterial concentrations in Pensacola Bay compared to Apalachicola Bay.

Weeks Bay has a much smaller watershed and estuarine area, nearly 2 orders of magnitude smaller than Apalachicola Bay or Pensacola Bay, and on the marine end exchanges with Mobile Bay estuary rather than the GOM proper, explaining the low mean salinity (5.7) we observed. The rate of water exchange between the 2 bays is strongly dependent on river discharge and wind forcing (Schroeder et al. 1990), but the mean freshwater residence
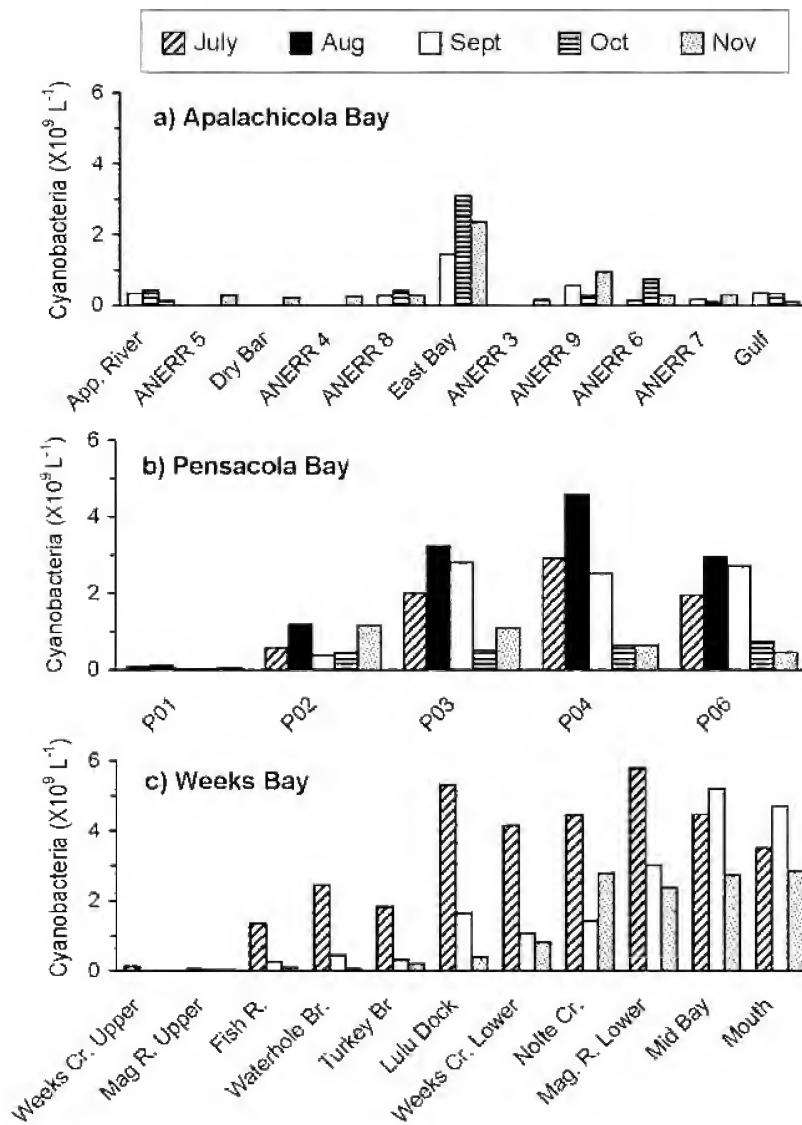
Figure 4. Cyanobacterial abundance at all sites and dates sampled during the study: a) Apalachicola Bay, FL; b) Pensacola Bay, FL; c) Weeks Bay, AL. Within each system, the sites were arranged in order of increasing mean salinity as listed in Table 2.

time is short (6 d) similar to Apalachicola Bay. It has high anthropogenic nutrient loading as evidenced by high freshwater DIN concentrations averaging 51 μM in the Fish River and 127 μM in the Magnolia River. It also has the highest baywide mean Chl $a$ concentrations and cyanobacterial abundances. Mean cyanobacterial abundances were about 350% higher than Apalachicola Bay and 50% higher than Pensacola Bay.

In this study, peak cyanobacterial abundances ranged from about 3 X $10^9$ L$^{-1}$ (Apalachicola Bay) to nearly 6 X $10^9$ L$^{-1}$ (Weeks Bay) and are among the highest reported in the literature (Table 4). Cyanobacteria have been enumerated in a wide range of estuarine and near-coastal environments, ranging from tropical (e.g., Phlips et al. 1999) to

northern latitude systems (e.g., Kuosa 1988). Cyanobacteria abundances in these systems vary considerably, but highest abundances always tend to occur during summer, and lower latitude systems tend to have higher peak abundances than higher latitude systems.

Because the time frame of this study was restricted to one summer-fall period, we acknowledge that the results may not be representative of longer-term patterns. As mentioned earlier, freshwater flows were below long-term averages, which likely caused higher salinities and lower water column stratification than expected to occur during more normal flow conditions. While interannual variation in such factors likely affect the location and extent of high cyanobacterial abundances, it seems clear from longer-
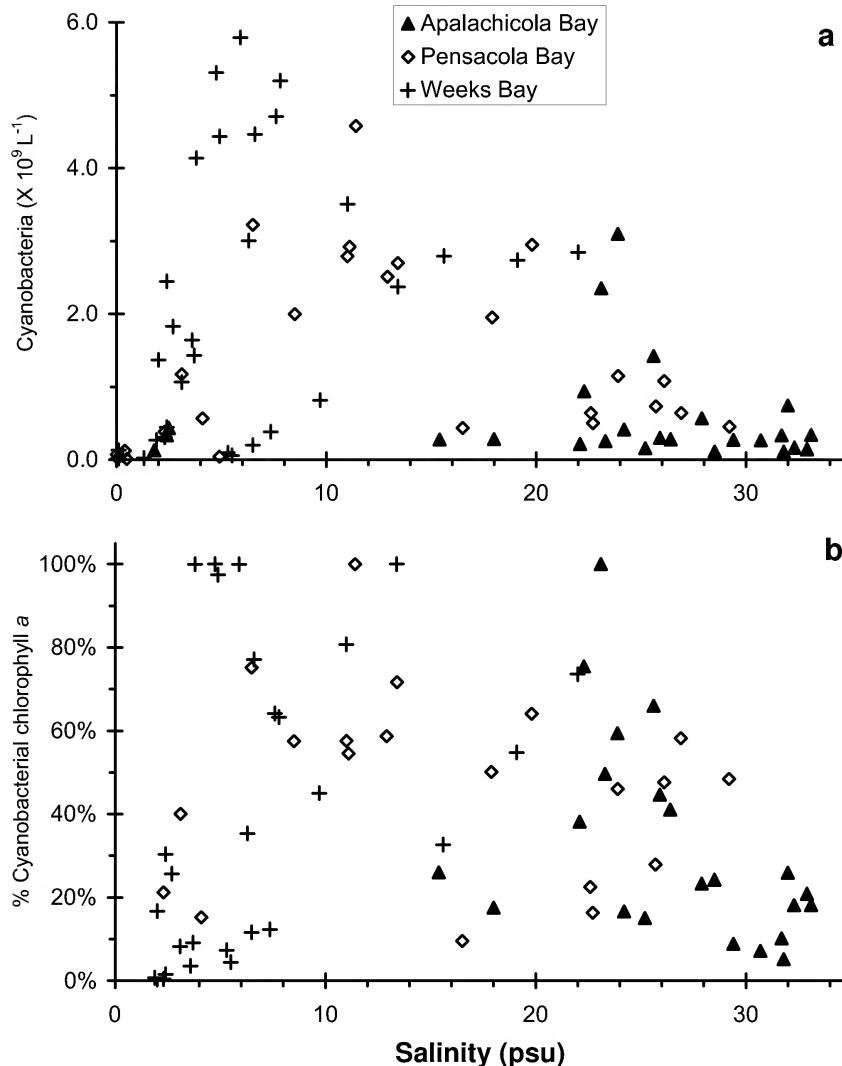
Figure 5. Distribution of cyanobacteria as a function of salinity: a) cyanobacterial abundance, and b) cyanobacterial percentage of bulk Chl *a*. The 3 estuaries are distinguished by different plot symbols.

term datasets (e.g., Marshall and Nesius 1996, Phlips et al. 1999, Murrell and Lores 2004) that high cyanobacterial abundances are a common summer-time feature of estuaries.

It is further clear from this study that cyanobacteria can be an important component of the phytoplankton community in these GOM estuaries, despite considerable variability in hydrology and anthropogenic impacts. Assuming a nominal cellular Chl *a* content, cyanobacteria contribute from 30 to 50% of the total Chl *a* in all 3 estuarine systems. This percentage agrees well with the 2+ year average of 43% reported for Pensacola Bay (Murrell and Lores 2004), and is among the highest reported in the literature. For example, in San Francisco Bay, cyanobacteria mean 15% (maximum 38%) of total Chl *a* (Ning et al. 2000). In the Neuse River estuary, cyanobacteria represented 18% of

total Chl *a* based on HPLC pigment analysis (Pinckney et al. 1998). In the York River estuary, pico-phytoplankton comprised 7% of Chl *a* over an annual cycle, peaking at 14% during summer (Ray et al. 1989). In the Kiel Bight, cyanobacteria contributed up to 52% of the total Chl *a* during summer (Jochem 1988), while in Southhampton estuary cyanobacteria contributed 10% or less to bulk Chl *a* (Iriarte and Purdie 1994). It should be noted that normalizing cyanobacteria to Chl *a* likely underestimates their true contribution to phytoplankton carbon biomass and productivity, given that cyanobacteria have relatively low chlorophyll content per unit of carbon compared to eukaryotic algae, particularly diatoms (MacIntyre et al. 2002).

One pattern noted by Iriarte and Purdie (1994) was that the relative importance of picoplankton appears to diminish with increasing trophic state, ultimately con-

## TABLE 4

**Peak abundances of cyanobacteria reported from various estuaries and inland seas. When available, temperature and salinity data at the time of collection are included and the month of the year the sample was collected.**

| Location | Temp °C | Salinity psu | Peak Abundance (cells X $10^6$L$^{-1}$) | Month | Reference |
|---|---|---|---|---|---|
| St. Lawrence River (Canada) | 21 | 0.1 | 17 | Jun | Bertrand and Vincent1994 |
| North Inlet (SC, USA) | NA | NA | 55 | Sep | Lewitus et al. 1998 |
| Southampton (UK) | 19–20 | 34 | 130 | Jul | Iriarte and Purdie 1994 |
| Yangtze River (China) | 25–30 | | 200 | Jul | Vaulot and Xiuren 1988 |
| Long Island Sound (NY, USA) | 24.3 | NA | 232 | Aug | Carpenter and Campbell 1988 |
| San Francisco Bay (CA, USA) | 22 | 20 | 234 | July | Ning et al.2000 |
| Gulf of Finland | 12–13 | 6 | 243 | Jun | Kuosa 1988 |
| Kiel Bight (Baltic Sea) | 22 | 14 | 260 | Jul–Aug | Jochem 1988 |
| York River Estuary (VA, USA) | 28 | 22 | 750 | Sep | Ray et al. 1989 |
| Chesapeake Bay (VA, USA) | 26 | NA | 920 | Jul | Affronti and Marshall 1994 |
| Chesapeake (MD & VA, USA) | NA | NA | > 2000 | Jul | Marshall and Nesius1996 |
| Apalachicola Bay (FL, USA) | 25 | 24 | 3100 | Sep | This Study |
| Pensacola Bay (FL, USA) | 29 | 11 | 4600 | Aug | Murrell and Lores 2004, This Study |
| Weeks Bay (AL, USA) | 30 | 6 | 5800 | Jul | This Study |
| Florida Bay (FL, USA) | NA | 35 | > 5000 | Oct | Phlips et al. 1999 |
| Mississippi River Plume (USA) | NA | 8 | > 5000 | Jul | Dortch 1998 |

tributing < 5% when Chl $a$ concentrations exceed 5 μg L$^{-1}$. In this study, cyanobacteria appeared to dominate the phytoplankton well beyond this 5 μg L$^{-1}$ threshold. The estuarine sites with the smallest cyanobacterial contribution (excluding freshwater sites) were the highly eutrophic sites (e.g. Weeks Bay) where total Chl $a$ concentrations exceeded 100 μg L$^{-1}$. Instead, phytoplankton at these sites were comprised of small diatoms (up to 6 X $10^7$ L$^{-1}$) and cryptophytes (up to 2.6 X $10^7$ L$^{-1}$). However, such highly eutrophic conditions are relatively rare in GOM estuaries, and seasonal maxima for Chl $a$ more typically range from 10 to 20 μg L$^{-1}$ (Pennock et al. 1999). In this range, the potential for cyanobacteria to dominate the phytoplankton is quite likely, given that an abundance of 5 X $10^9$ L$^{-1}$ corresponds to 17 μg L$^{-1}$ Chl $a$ (assuming 3.4 fg Chl $a$ cell$^{-1}$). Therefore, data from this and related studies (e.g., Phlips et al. 1999, Murrell and Lores 2004) appear to challenge the generalized pattern observed by Iriarte and Purdie (1994), showing that cyanobacteria can be dominant in GOM estuaries and can represent nearly 100% of the Chl $a$, especially during summer.

While there are several reports of cyanobacterial abundances in estuaries, cyanobacterial growth rates and productivity are more rarely quantified. However, studies conducted in several estuaries, including Chesapeake Bay (Affronti and Marshall 1994), Long Island Sound (Carpenter and Campbell 1988), the South China Sea

(Agawin et al. 2003), and Santa Rosa Sound (Juhl and Murrell in press) have consistently found that peak specific growth rates range from 1 to 1.5 d$^{-1}$ (1.4 to 2.2 divisions d$^{-1}$). One consistent finding in these and related studies is a strong temperature-dependence on cyanobacterial growth, being repeatedly noted in estuarine (Carpenter and Campbell 1988, Ray et al. 1989, Iriarte and Purdie 1994, Juhl and Murrell in press) and oceanic environments (Li 1998). Based on these observations, it is clear that estuarine cyanobacteria actively grow during warm periods and significantly contribute to bulk productivity. Furthermore, given their characteristically low chlorophyll content relative to carbon (MacIntyre et al. 2002), cyanobacterial contribution to bulk phytoplankton productivity probably exceeds their contribution to bulk Chl $a$. Thus, cyanobacteria appear to be major mediators of carbon flow in subtropical estuarine systems and deserve further study to better quantify their role in estuarine productivity.

The size structure of the phytoplankton community has a profound influence on the pathways by which organic matter is transferred through aquatic food webs. Perhaps most importantly, cyanobacteria in the 1 to 2 μm size range cannot be directly consumed by mesozooplankton and demersal fish species. For example, Nival and Nival (1976) found that even naupliar stages of the ubiquitous genus *Acartia* was unable to efficiently collect and consume particles less than 3 μm in size. Similarly, Durbin and Durbin

(1975) found that the Atlantic menhaden (*Brevoortia tyrannus*), a major phytoplanktivorous fish in estuaries, was unable to consume phytoplankton less than 13–16 µm in size. So the route by which cyanobacteria become available to higher trophic levels requires one or more intermediate trophic steps (i.e. the microzooplankton), with respiratory losses of carbon and energy at each step. The existence of such trophic linkages has been demonstrated, in particular between cyanobacteria and microzooplankton (Caron et al. 1991, Ayukai 1992, Lessard and Murrell 1998, Juhl and Murrell in press), and between microzooplankton and mesozooplankton (Lonsdale et al. 1996, Sipura et al. 2003). However, the inefficiency of such indirect pathways, when compared to more direct pathways, constrains the degree to which cyanobacteria can ultimately support production of top predators.

In summary, this study found high abundances of chroococcoid cyanobacteria in 3 estuaries along the northeastern GOM. Cyanobacterial abundances peaked in the oligohaline reach of each system and appeared to positively covary with the degree of eutrophication. While cyanobacteria have long been known to play a dominant role in oceanic environments, their role in estuaries is not as well understood. This study adds to a small but growing body of literature suggesting that cyanobacteria can be dominant in estuaries, which has broad implications for how primary production is transferred to higher trophic levels. Future studies should consider the potential role of cyanobacterial dominance on various topics ranging from fish productivity to eutrophication effects.

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Ecology of the Mayan Cichlid, *Cichlasoma urophthalmus* Günther, in the Alvarado Lagoonal System, Veracruz, Mexico

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# ECOLOGY OF THE MAYAN CICHLID, *CICHLASOMA UROPHTHALMUS* GÜNTHER, IN THE ALVARADO LAGOONAL SYSTEM, VERACRUZ, MEXICO

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*ABSTRACT* The Mayan cichlid, *Cichlasoma urophthalmus*, has a wide distribution in southeastern Mexico where it inhabits rivers and coastal lagoons. In the Alvarado lagoonal system, Veracruz, it is distributed towards the north in Camaronera Lagoon. The Mayan cichlid shows an affinity for oligohaline to mesohaline sites with submerged vegetation, well-oxygenated, deep, and transparent water. The major abundance and biomass of this species was obtained during December to February. The diet of Mayan cichlids consists principally of plant detrital material and algae. Length-frequency distribution shows 2 size classes during both the dry and rainy seasons, corresponding to reproductive fish and young of the year; during the nortes season there is only one modal size class of fish between 60–100 mm SL. Individuals with developed gonads are found throughout the year, although most reproductive adults are found between April and December. The highest Gonadosomatic Index (GSI) values coincided with the peak in reproductive activity between May and July. The fecundity ranged from 1,556–3,348 eggs/female, and there was no relationship between female size and fecundity.

*RESUMEN* El cíclido maya, *Cichlasoma urophthalmus,* tiene una distribución amplia en el sureste de México, donde habita ríos y lagunas costeras. En el sistema lagunar de Alvarado, Veracruz, esta especie se distribuye hacia el norte principalmente en la Laguna Camaronera. Esta especie muestra afinidad por sitios oligohalinos y mesohalinos con vegetación sumergida, bien oxigenados, profundos y de aguas transparentes. La mayor abundancia y biomasa de *C. urophthalmus* fue obtenida durante Diciembre a Febrero. La dieta del cíclido maya consistió principalmente de detritus vegetal y algas. La distribución frecuencia-longitud mostró dos clases de talla durante las temporadas de secas y lluvias, correspondientes a individuos reproductores y menores de un año; durante la temporada de nortes se encontró solo una clase de talla modal entre 60–100 mm LS. Los individuos con gónadas desarrolladas se encontraron a lo largo del año, aunque los adultos más reproductivos se encontraron entre Abril y Diciembre. El valor más alto del Índice Gonadosomático (IGS) coincidió con el pico de actividad reproductiva entre Mayo y Julio. La fecundidad se registró entre 1,556–3,348 huevos/hembra y no hubo relación entre la talla de las hembras y su fecundidad.

## INTRODUCTION

The family Cichlidae is of freshwater origin with about 1300+ species worldwide (Nelson 1994). Members of this family are notorious for their capacity to colonize diverse habitats such as rivers, estuaries, lakes and coastal lagoons, representing a notable adaptation to a wide range of physical, chemical and biological environmental variables. The genus *Cichlasoma* comprises 118 species from the New World (Alvarez del Villar 1970, Astorqui 1971, Kullander 1983, 2004). At least 39 species are found in Mexico, where *Cichlasoma* is the most diverse genus of the freshwater ichthyofauna (Alvarez del Villar 1970). The Mayan cichlid, *Cichlasoma urophthalmus,* occurs in fresh and brackish waters of the Atlantic watersheds from the Rio Coatzacoalcos basin southward through Mexico, including the Yucatan Peninsula and Isla Mujeres, into Belize, Guatemala, Honduras and Nicaragua (Miller 1966, Martínez-Palacios and Ross 1992, Greenfield and Thomerson 1997).

The Mayan cichlid has been recorded in the State of Veracruz in the southern Gulf of Mexico (GOM) in the Pánuco, Sarabia, Chachalacas, Papaloapan, Coatzacoalcos, and Achotal Rivers. In the Alvarado lagoonal system, 3 genera and at least 7 species of cichlids have been reported, with the Mayan cichlid being the species with highest abundance and greatest ecological importance among the freshwater species of the system (Chávez-López 1998). Mayan cichlids have been reported in river-lagoonal systems associated with Términos Lagoon, Campeche (Amezcua-Linares and Yáñez-Arancibia 1980). They have also been reported from the Champotón River basin, Campeche, the Grijalva-Usumacinta River basin, Tabasco, the Yucatan Peninsula and Isla Mujeres in Quintana Roo (Miller 1966, Reséndez-Medina 1981, Caso-Chávez et al. 1986, Martínez-Palacios 1987), and on barrier reefs in Belize (Greenfield and Thomerson 1997).

In Mexico, the Mayan cichlid is exploited commercially in the artesanal fishery and has a potential for aquaculture in freshwater areas (Miller 1966, Martínez-
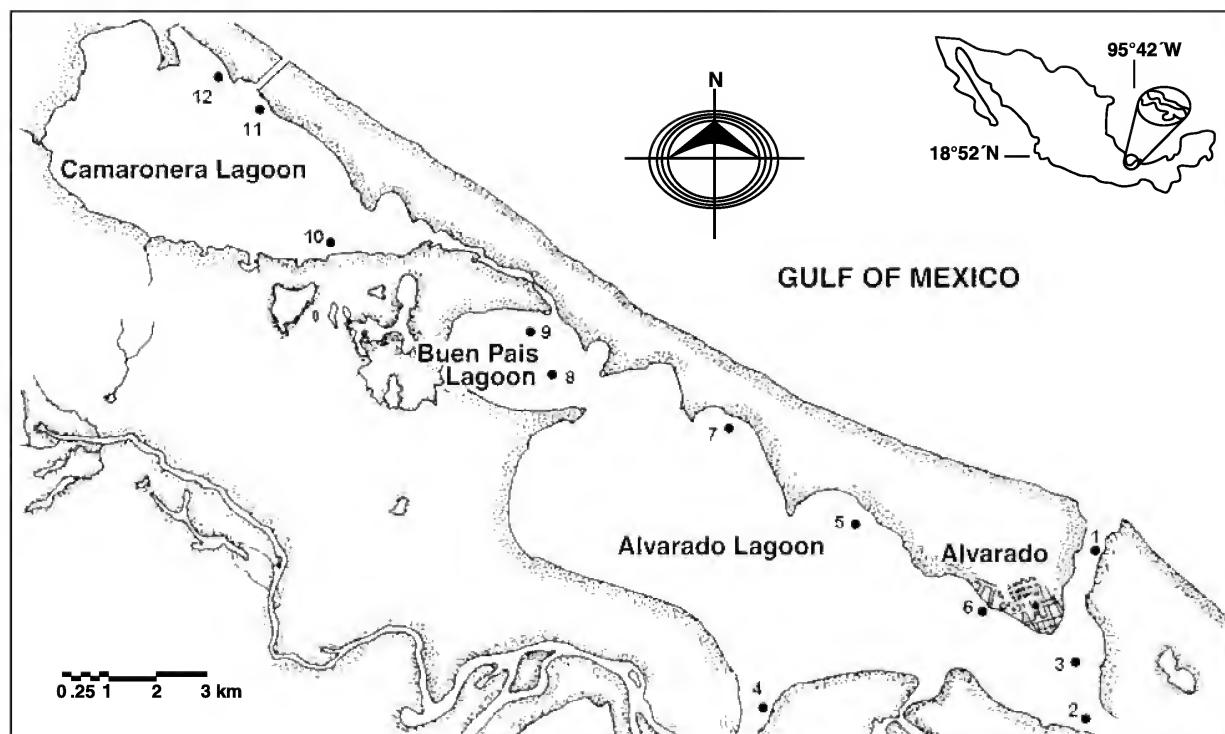
Figure 1. Sampling stations during June 2000–July 2001 in the Alvarado lagoonal system, Veracruz, Mexico. Inset map shows approximate geographic location of the study area.

Palacios and Ross 1992, Ross and Beveridge 1995). In the local markets, it has been preferred over introduced species such as tilapias, and the price was 20–40% higher than that for introduced species in Campeche (Résendez-Medina 1981), where this species has been sold along with marine species.

It has been shown that the Mayan cichlid is extremely adaptable throughout its range and that aspects of its life history vary depending on environmental conditions; for example, see Loftus (1987), Martínez-Palacios et al. (1990), Martínez-Palacios and Ross (1992), Faunce and Lorenz (2000), Faunce et al. (2002), and Bergmann and Motta (2004). The purpose of the present study is to provide additional information on the biology and reproduction of Mayan cichlids in the Alvarado lagoonal system in Veracruz, Mexico.

## MATERIALS AND METHODS

### Field collections and processing of specimens

Mayan cichlids were collected on 12 dates from June 2000 to July 2001 at 12 stations in the Alvarado lagoonal system, Veracruz, Mexico (Figure 1). Physicochemical data taken at each station included depth (cm), Secchi transparency (cm), salinity (psu), temperature (°C), dis-

solved oxygen (mg/l), and turbidity (NTU); percent submerged aquatic vegetation (SAV) cover was estimated visually. For analysis, monthly collections were divided into seasons following Raz-Gusmán et al. (1992), where the dry season was March through June, the rainy season was July through October, and the nortes season was November through February.

Fish were captured at each station using a single haul of a 30 m long x 2 m high seine constructed with 6.35 mm mesh. Total area sampled with each seine haul was 756.25 m². Fish were fixed in 10% buffered formalin and also injected in the abdominal cavity in the field to retard the digestive process. Fish were washed in fresh water after one week in formalin and then stored in 70% ethanol. Species identification followed Alvarez del Villar (1970) and Reséndez-Medina (1981). Each fish was measured to the nearest mm standard length (SL), weighed (WW) to the nearest g, and the gonads were removed and weighed to the nearest 0.1 g. Seasonal feeding of Mayan cichlids was determined by analyzing the stomach contents of 35 fish from each season. Stomach contents were separated to the lowest taxonomic level possible, weighed (0.01g) and analyzed according to the gravimetric method (Hyslop 1980).

Gonadal maturity classes were established by extract-

ing a 1 mm thick portion of the ovaries or testes. Semi-permanent preparations were examined microscopically to determine the class of gonadal development following Murphy and Taylor (1990). The Gonadosomatic Index (GSI) was determined using the formula GSI = [gonad weight/(total wet weight – gonad weight)/100]. The total number of vitellogenic oocytes (> 0.42 L x 0.30 W mm) in the ovaries of mature females (87 mm–146 mm SL, 28.6–66.7 g) were counted to estimate fecundity.

## Statistical Analysis

The relationships between SL and WW of male and female Mayan cichlids and between fecundity and female SL and weight were examined using linear regression analyses. A regression of GSI vs. body weight was used to verify that GSI was an appropriate index of spawning preparedness. The GSI data were arc sine transformed and then compared by gender across months with ANOVA. If a significant $F$-value was determined, pairwise Sidak tests were used to separate mean values. A Mann-Whitney U-test was used to compare each of seven physicochemical variables between stations with and without Mayan cichlids. Diet was compared among seasons using the Bray-Curtis similarity coefficient $C_Z$, with 0 = most dissimilar diets and 1 = identical diets (Marshall and Elliott 1997). Differences in length-frequency distributions among seasons were compared with pairwise Kolgomorov-Smirnov 2-sample Chi-square tests.

The relationship between the physicochemical variables and Mayan cichlid abundance was also examined using Principal Component Analysis (PCA) in a 2 step procedure (Peterson and VanderKooy 1997). First, the stations were ordered based on the seven physicochemical variables with PCA of the correlation matrix, with varimax rotation to maximize the loading results. A Scree Test was used to determine the number of components, and stan-

dardized scores of the factors were plotted for each station/month period against the meaningful components. Second, these station/month coordinates were coupled with the abundance of the Mayan cichlid for that specific collection station. Any variable with a correlation ≥ 0.50 was considered when interpreting a component. All calculations were made using SPSS software (Versions 10.0 or 11.5, Chicago, IL) and the results were considered significant if $P < 0.05$.

## RESULTS

### Abundance

Mayan cichlids were captured in 59 of 128 collections (46.1%) from the 12 stations in the Alvarado lagoonal system. The frequency of capture was similar among the stations except for the Blanco River station (# 4), where the Mayan cichlid was taken during only 4 of 12 collections, and in the estuarine zone of Papaloapan River (stations 1–3) where the species was not collected.

Adult Mayan cichlids ranged from 87 to 146 mm SL and had the greatest abundance in Camaronera Lagoon ($n$ = 672, representing 52.7% of the total fish caught) and in the Aneas, Arbolillo, and Buen Pais stations (# 6,7,8,9) on the internal margin of the barrier separating the lagoon from the ocean. Abundance was greatest between November to February (nortes season) and lowest in April. Mayan cichlids were most abundant at stations containing SAV (Table 1); such as stations 10 and 11 in Camaronera Lagoon ($n$ = 639, 49.4%), stations 8 and 9 in Buen Pais Lagoon ($n$ = 200, 15.2%), and at the Arbolillo station (# 7) in Alvarado Lagoon ($n$ = 216, 16.4%).

### Habitat Relationships

The Mayan cichlid showed affinity for mesohaline stations, which were most common during the nortes and dry seasons and least common in the low salinity rainy sea-

## TABLE 1

**Abundance of *Cichlasoma urophthalmus* expressed as a percentage of the individuals collected, as percentage in stations with < 50% coverage with submerged aquatic vegetation (SAV), as percentage in stations with about 50% coverage with SAV, and as a percentage in stations with > 50% coverage with SAV. Sampling stations correspond to stations on Figure 1.**

| | Sampling Stations | | | | | | | | | | | |
| | 12 | 11 | 10 | 9 | 8 | 7 | 5 | 6 | 4 | 2 | 3 | 1 | TOTAL |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Overall abundance | 3.3 | 32.4 | 17.0 | 15.2 | 7.4 | 16.4 | 0.5 | 6.3 | 1.5 | | | | 100.0 |
| Stations with | | | | | | | | | | | | | |
|     SAV < 50% | 3.3 | | | | 0.5 | 15.4 | 0.4 | 4.3 | 1.5 | | | | 25.4 |
|     SAV = 50% | | 0.8 | 16.9 | | | | | 1.2 | | | | | 18.9 |
|     SAV > 50% | | 31.6 | 0.1 | 15.2 | 6.9 | 1.0 | 0.1 | 0.8 | | | | | 55.7 |

## TABLE 2

**Comparison of physicochemical factors ($\bar{x} \pm s$) between habitats with with and without *Cichlasoma urophthalmus*. Significant difference\* ($P < 0.05$) determined by a Mann-Whitney U test.**

| Parameter | All stations | Stations with *C. urophthalmus* | Stations without *C. urophthalmus* |
|---|---|---|---|
| Submerged Vegetation (%)* | 41.35 ± 34.47 | 57.24 ± 30.59 | 30.12 ± 32.74 |
| Depth (cm) | 99.11 ± 82.90 | 83.36 ± 30.96 | 110.25 ± 104.00 |
| Secchi transparency (cm) | 43.9 ± 21.93 | 45.11 ± 16.35 | 43.05 ± 25.22 |
| Salinity (psu)* | 5.72 ± 5.70 | 7.13 ± 5.32 | 4.72 ± 5.79 |
| Dissolved Oxygen (mg/l)* | 9.39 ± 1.76 | 9.72 ± 1.70 | 9.16 ± 1.77 |
| Temperature (°C) | 27.8 ± 2.92 | 27.96 ± 2.92 | 27.7 ± 2.94 |
| Turbidity (NTU) | 17.35 ± 14.86 | 15.63 ± 10.63 | 18.6 ± 17.20 |

son. Stations with Mayan cichlids exhibited greater SAV ($Z = 4.42$, $P < 0.001$), higher salinity ($Z = 3.16$, $P < 0.001$), and slightly higher dissolved oxygen ($Z = 1.78$, $P = 0.076$) than stations without Mayan cichlids (Table 2). There were no differences among the other variables measured (all $P > 0.05$). The SAV was composed mainly of *Ruppia maritima* with various percentages of the algae *Gracillaria verrucosa* and *Rhizoclonium hieroglyphicum* in Camaronera and Buen Pais Lagoons; other stations with SAV had only beds of *R. maritima*. There were no statistically significant differences in temperature, depth, transparency, and turbidity between stations where Mayan cichlids were present vs absent ($P > 0.05$; Table 2), but Mayan cichlids tended to occur at shallower and less turbid stations (Table 2).

The PCA analysis extracted 3 axes that accounted for 66.26% of the total variation in the physicochemical data (Table 3). The first component represents transparency (+), salinity (+), dissolved oxygen (+) and turbidity (-). The second component represents SAV (-) and depth (+), and

the third component represents water temperature (+). The analysis indicates that stations of shallow depth, greater SAV (*Ruppia maritima*) cover, high salinity, high dissolved oxygen, high transparency, and low turbidity had the greatest numbers of Mayan cichlid captured (Figure 2). These stations were located in Camaronera Lagoon, usually during the nortes and dry seasons.

### Size Distribution

Length-frequency histograms were constructed for each season (Figure 3). There were clear bimodal size distributions for the rainy season (20–40 mm and 101–120 mm SL) as well as the dry season (1–40 mm SL and 81–120 mm SL), indicating numerous small Mayan cichlids. In contrast, the highest frequencies in the nortes season corresponded to 60–100 mm and 120–160 mm SL, with no small fish being collected. Comparison of SL size distributions among seasons, pooled by gender, indicated that there was no significant difference between dry and rainy seasons ($Z = 1.322$, $P = 0.061$) or between rainy and nortes seasons ($Z = 0.685$, $P = 0.737$). However, the dry and nortes season SL size distributions were different ($Z = 1.958$, $P = 0.001$).

Log$_{10}$ SL vs log$_{10}$ WW for all females was significant ($F = 1600.59$, $r = 0.94$, $P < 0.001$, $n = 210$) and explained by log$_{10}$ WW = $-1.460 + 2.984$ log$_{10}$ SL. For males, log$_{10}$SL vs log$_{10}$ WW was significant ($F = 1938.99$, $r = 0.96$, $P < 0.001$, $n = 168$), and explained by log$_{10}$ WW = $-1.240 + 2.752$ log$_{10}$ SL.

### Diet

The Mayan cichlid was predominately herbivorous in the Alvarado lagoonal system (Table 4), with a total of 19 food types identified. All fish had plant material in the stomach, and the percentages varied by season from a low of 74.41% in the dry season to a high of 98.3% in the

## TABLE 3

**Physicochemical variables correlated with the 3 principal components with eigenvalues > 1. The percent of variance explained by each component is in parenthesis. Variables with correlations > 0.50 are used in identifying the components.**

| | PC-I (30.15%) | PC-II (20.07%) | PC-III (16.04%) |
|---|---|---|---|
| Depth (cm) | 0.084 | 0.858 | 0.121 |
| Submerged Vegetation (%) | 0.185 | – 0.660 | 0.376 |
| Secchi transparency (cm) | 0.683 | 0.414 | 0.191 |
| Salinity (psu) | 0.737 | – 0.192 | 0.008 |
| Dissolved Oxygen (mg/l) | 0.688 | – 0.093 | – 0.350 |
| Temperature (°C) | 0.018 | – 0.039 | 0.902 |
| Turbidity (NTU) | – 0.753 | – 0.067 | – 0.155 |
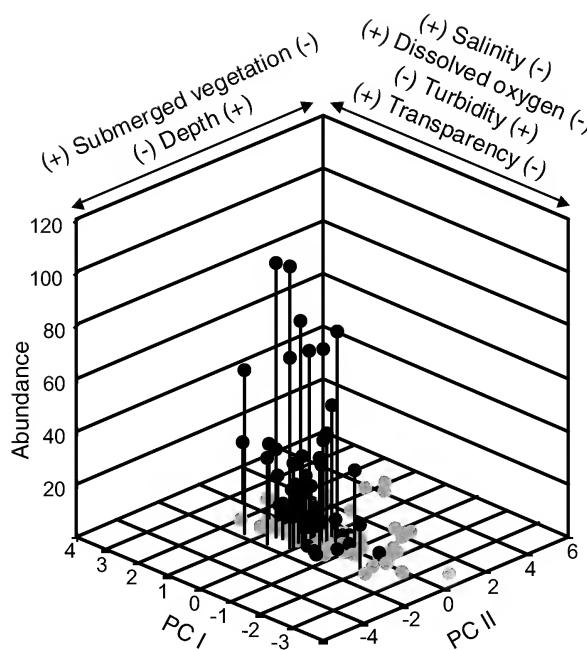
Figure 2. Three-dimensional plot of the stardardized factor scores for the stations and months of collection and the abundance of *Cichlasoma urophthalmus* arranged on principal components I and II based on seven physicochemical variables. Black lollypops are where Mayan cichlids were collected, whereas gray lollypops are where no Mayan cichlids were collected.

nortes season. Mayan cichlids supplemented its herbivorous diet with 18 other food types (2.12% of the diet) in the rainy season, 4 other food types (1.83% of the diet) in the nortes season, and 6 other food items (26.6% of the diet) in the dry season. The dry season diet was unique in that it was composed of a number of animal taxa, particularly mollusks (20.1%), crustaceans (3.75%) and fish scales (2.07%). The diets of Mayan cichlid were most similar between the rainy and nortes seasons ($C_z = 0.9816$), There was reduced diet similarity between the rainy and dry seasons ($C_z = 0.6716$) and between the nortes and dry seasons ($C_z = 0.6705$).

## Reproduction

Males, females, and juvenile Mayan cichlids were found in all collections in the Alvarado lagoonal system. Overall, the sex ratio of mature individuals was 1.16:1 (female:male). Gonadal recrudescence was first observed in individuals > 100 mm SL in April, although individuals as small as 60 mm SL showed gonadal development in July.

A comparison of GSI and gonad-free wet weight for

females ($r^2 = 0.107$, $P < 0.001$, $n = 314$) and males ($r^2 = 0.068$, $P < 0.001$, $n = 247$) showed that while there is a significant, positive relationship between GSI and body weight, GSI explains $\leq 10.7\%$ of the variation in weight. Thus, GSI can be used as an index of spawning preparedness for this species. Female GSI varied significantly across sampling dates ($F_{11,314} = 12.177$, $P < 0.001$). The GSI indicates maximal ovarian development from May–July, with a GSI peak in May (Figure 4). Elevated GSI values were also seen in June–July 2000, verifying that maximal female reproductive activity occurs at the end of the dry season (May–June). However, there was a small peak in female GSI in December. The highest GSI values in May and June corresponded to females 120–160 mm SL. In contrast, male GSI values were significantly different over time ($F_{11,247} = 3.062$, $P < 0.001$) and showed much greater variability over the season than did those of females (Figure 4). Male GSI peaks occured in May–July in both years and in January 2001, similar to peaks seen in females. The large variation in GSI most likely indicates that individuals were in all stages of gonadal development each month, suggesting a protracted reproductive season.

During all months, individuals with undifferentiated and immature or regressed (stage I) ovaries were collected, and these individuals made up the majority of the females collected (Figure 5). Fish with ovaries in stages II and III were captured from May–July and December–February, while reproductive individuals (stage IV) were captured from May–July and December, with the greatest percentages found in June and July in both 2000 and 2001 (Figure 5).

Females ranging from 87–145 mm SL had fecundity
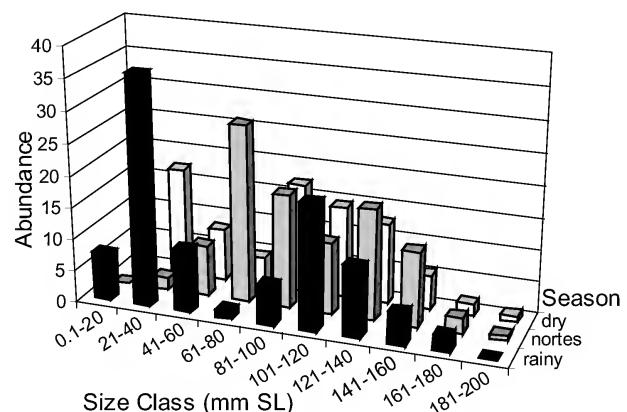


Figure 3. Size distribution of *Cichlasoma urophthalmus* by season in the Alvarado lagoonal system, Veracruz, Mexico. Dry season was March through June; the rainy season was July through October; and the nortes season was November through February.

TABLE 4

**Seasonal diet composition (% weight) of *Cichlasoma urophthalmus* in Alvarado lagoonal System.**

| Food items | Rainy season | Nortes season | Dry season |
|---|---|---|---|
| Plant organic matter | 97.28 | 98.17 | 67.16 |
| Fish scales | 0.62 | 0.54 | 2.07 |
| Algae | 0.58 | 0 | 0.48 |
| Tanaidacea | 0.57 | 0 | 0 |
| *Ruppia maritima* | 0.48 | 0.20 | 5.77 |
| Annelids | 0.13 | 0 | 0 |
| Crustacea | 0.10 | 0.084 | 3.75 |
| Insects | 0.074 | 0 | 0.67 |
| Nematodes | 0.06 | 0 | 0 |
| Molluscs | 0.048 | 0.99 | 20.10 |
| Animal organic matter | 0.016 | 0 | 0 |
| Hydrobiidae | 0.007 | 0 | 0 |
| Amphipoda | 0.005 | 0 | 0 |
| Isopoda | 0.002 | 0 | 0 |
| Diptera | 0.001 | 0 | 0 |
| Fish eggs | 0.001 | 0 | 0 |
| Acari | 0.0002 | 0 | 0 |
| *Fisaria* sp. | 0.0001 | 0 | 0 |
| Cladocera | 0.0001 | 0 | 0 |

values from 1,556 to 3,348 eggs/female. There was no correlation between fecundity and SL of females (Fecundity = 1,916.92 + 2.780 SL; $r = 0.0835$, $n = 14$, $P = 0.74$), as small females often had a greater number of oocytes compared with large females.

## DISCUSSION

Mayan cichlids were closely associated with habitat characterized by SAV and salinities between 3 and 13 psu in the Alvarado lagoonal system. This explains why the majority of the Mayan cichlids collected were taken in the nortes season and greater abundance was observed in Camaronera and Buen Pais Lagoons. Mayan cichlids were absent from the three Papaloapan River stations that have zero or low salinity throughout the year. The results of the present study agree with Caso-Chávez et al. (1986), who reported a greater number of Mayan cichlids in zones influenced by the ocean and with the presence of seagrass in Terminos Lagoon, Mexico. Mayan cichlids are also reported to have the greatest abundance in salinities up to 25 psu in the Mexican Caribbean (Martínez-Palacios and Ross 1992) and Florida (Faunce and Lorenz 2000). In fact, water temperature and salinity are not likely to limit their range in non-native habitat types in south Florida except in really cold winters, because at 25 °C, salinity tolerance is
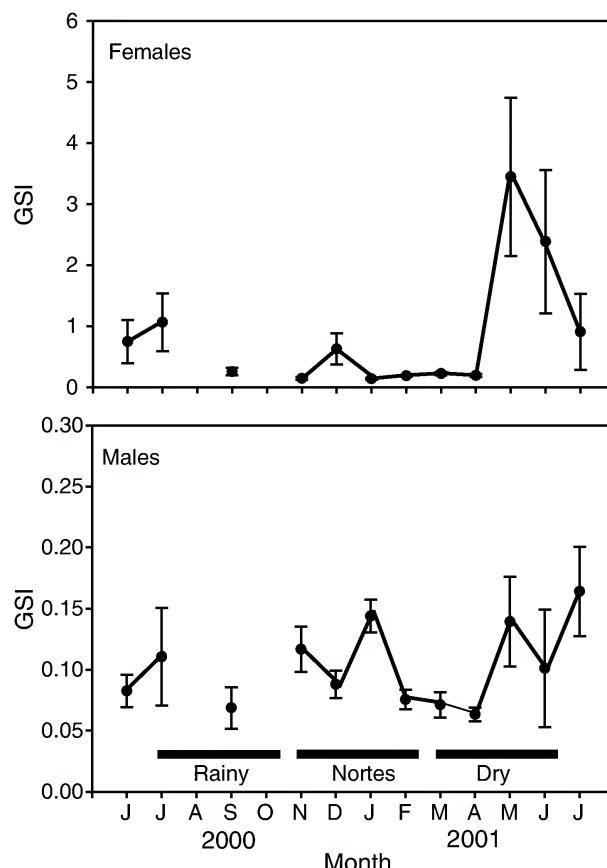


Figure 4. Plot of the gonadosomatic index (GSI; $\bar{x} \pm s_{\bar{x}}$) by month of female ($n = 314$) and male ($n = 247$) *Cichlasoma urophthalmus* from the Alvarado lagoonal system, Veracruz, Mexico. No collections were made in August and October 2000.

> 37 psu (Stauffer and Boltz 1994).

There were 2 size class distributions of the Mayan cichlid documented in the Alvarado lagoonal system. During the nortes season, mainly pre-adults and adults (61–160 mm SL) were captured, as has been reported in other Mexican locations (Caso-Chávez et al. 1986, Martínez-Palacios and Ross 1992). In contrast, the size class distribution was bimodal during the dry season, representing both juvenile recruits (10–20 mm SL) and reproductive adults (81–120 mm SL). The bimodal pattern shifted to larger sizes in the rainy season, with fish between 40–60 mm SL being most numerous, followed by a cohort of fish between 140–160 mm SL. We propose 2, nonexclusive explanations for the lack of larger Mayan cichlids collected in our study. First, the populations may suffer from overfishing as has been documented in the Celestún Lagoon, Mexico (Martínez-Palacios and Ross 1992); the minimum commercial size for this species in the Alvarado lagoonal system is 150 mm SL. Second, larg-
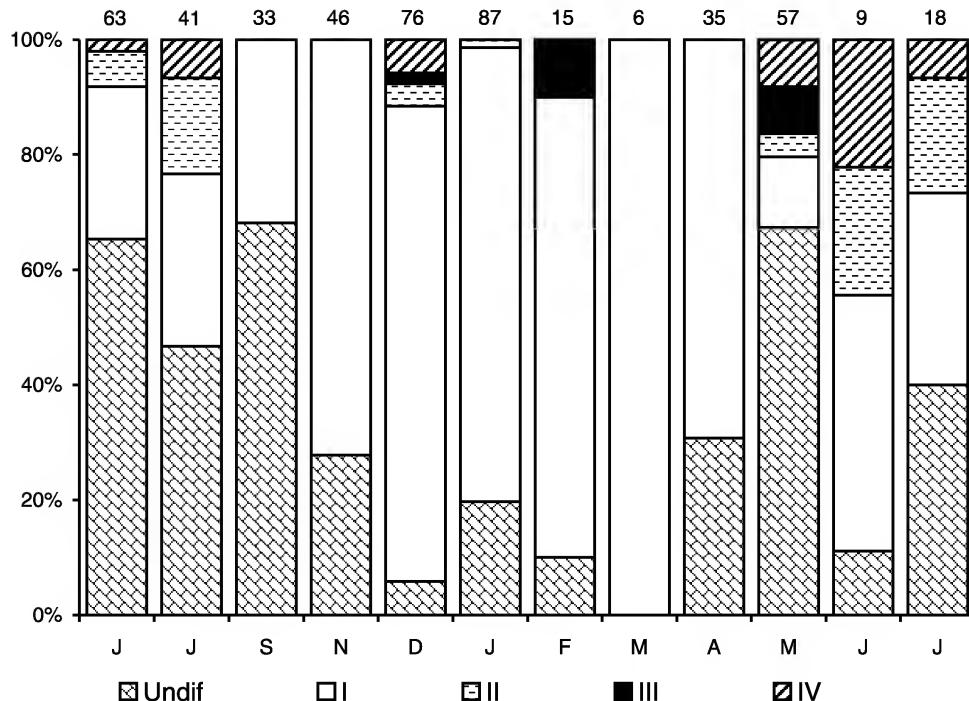
**Figure 5. Monthly ovarian classes of** *Cichlasoma urophthalmus* **in the Alvarado lagoonal system, Veracruz, Mexico. Individual sample sizes by month are provided above each histogram. No collections were made in August and October 2000.**

er fish have been shown to migrate to deeper water habitats in Florida systems (Faunce et al. 2002), and we suggest our inability to collect larger Mayan cichlids is due in part to our shallow water sampling techniques and we used relatively small seines. In fact, we did not collect individuals in spawning or post-spawning condition during this study, which may suggest that Mayan cichlids select other sites in the lagoonal system or immediately offshore to complete their reproduction. For example, Mayan cichlids have been observed breeding in seawater over sand on the barrier reef behind St. George Cay (Greenfield and Thomerson 1997), a different habitat type than those sampled in the Alvarado lagoonal system. Mayan cichlids with large ($\bar{x}$ = 1.72 mm) diameter oocytes were captured in Celestún Lagoon, Mexico, where mean salinity ranged from 16–24 psu (Martínez-Palacios and Ross 1992); these diameters are much higher than any value measured in the Alvarado lagoonal system.

The diet of Mayan cichlids was principally herbivorous but varied seasonally, most likely in response to prey availability. Although plant material was the main food item, diet in the dry season was composed of a considerable portion of crustaceans, insects, and mollusks, similar to findings by Chávez-Lopez (1998). Mayan cichlids collected in *Thalassia testudinum* grassbeds in Terminos Lagoon, Mexico, were mainly ingesting plant and detrital

matter, with sponges and cirripeds as incidental food (Caso-Chávez et al. 1986). In contrast, Mayan cichlids (96–200 mm SL) in the Celestún Lagoon, Mexico, were classified as carnivorous, feeding mainly on small invertebrates (palaemonid and penaeid shrimp) with little algae or seagrass (Martínez-Palacios and Ross 1988). Finally, Bergmann and Motta (2004), based on diet and trophic morphology, indicated that Mayan cichlids in southern Florida were generalists, feeding on fish and snails, and that being generalist and opportunistic feeders enhanced its colonization success in non-native environments.

It appears the reproductive season is more prolonged in coastal Mexican lagoons, likely caused by factors such as temperature and day length (Noakes and Balon 1982, Munro et al. 1990). Although Mayan cichlids have a protracted reproductive period in the Alvarado lagoonal system, we found females with mature eggs only between May and July. Caso-Chávez et al. (1986) reported that reproductive activity was maximal in June and no reproductive females were collected after September in Terminos Lagoon, Mexico. Martínez-Palacios and Ross (1992) indicated that the reproductive season began in mid-April and ended by mid-November in the Yucatan Peninsula. In contrast, the reproductive season in Florida appears to occur only in April and May (Loftus 1987, Faunce and Lorenz 2000). The reproductive season in

Mexico (Martínez-Palacios and Ross 1992) stopped when temperatures dropped below 24 °C, from late-November to March, whereas in Florida, reproduction stopped in October at 23 °C (Faunce and Lorenz 2000). In the Alvarado lagoonal system, we did not find mature females in the coldest months of the year (January and February) when water temperature had decreased to 23 °C.

Our data are comparable with all other reports that sexual maturation occurs by 100 mm SL in Mayan cichlids. In the Yucatan Peninsula, Mexico, the minimum size for female maturity is 102 mm SL, enabling females to reproduce during their first spring as they approach their first birthday (Martínez-Palacios and Ross 1992). Females in Terminos Lagoon, Mexico, reached sexual maturity at 60 mm SL (Caso-Chavez et al. 1986). We also found mature females as small as 60 mm SL but only in July toward the end of the reproductive season. In contrast, Mayan cichlids from northern locations in Florida reach 50% sexual maturity at 127.2 mm SL (Faunce and Lorenz 2000), suggesting there may be latitudinal variation in size at maturity as reported for other cichlid species (Turner and Robinson 2000).

Surprisingly, we found no relationship between female size and fecundity for Mayan cichlids in the Alvarado lagoonal system, although a significant positive relationship has been previously reported for this species in Celestún Lagoon, Mexico (Martínez-Palacios and Ross 1992). The small sample size for fecundity estimates may contribute to the lack of a significant relationship. Even when a significant relationship is seen between fish size and fecundity, size explains only 33% of the variation in fecundity (Martínez-Palacios and Ross 1992). Nonetheless, the range of fecundity values we obtained overlap the low end of the range reported by Martínez-Palacios and Ross (1992; 2085–6615 ova/female, 113–198 mm SL) and were based on smaller fish (87–146 mm SL) than those in the Yucatan.

Camaronera Lagoon, the northern part of the system, had the highest salinity between April and June (dry season), when the majority of reproductive activity occurs and when nest construction and parental care occurs in Florida populations (Faunce and Lorenz 2000). The rainy season begins in July and the salinity decreases to 5 psu in this zone as the water levels begin to increase. This coincides with the termination of parental care and the migration of juveniles to other areas to find lower salinity and warmer temperatures (34 °C in Alvarado Lagoon). In the lower salinities common during the rainy season, juveniles are in an almost isotonic aquatic medium at salinities which facilitate the best growth of Mayan cichlids < 1 year old (Martínez-Palacios et al. 1990). Furthermore, the abun-

dance of adults decreases in the shallow areas of the lagoonal system during the rainy season, suggesting they may move to deeper areas with higher salinities.

In spite of the wide distribution of Mayan cichlids in the southeast of Mexico, until now little was known regarding the state of natural populations. Some populations of Mayan cichlids that inhabit cenotes (sinkholes) in the Yucatan Peninsula are considered species of special concern in Mexico (Diario Oficial de la Federación 2002). However, Mayan cichlids were suggested as a native aquaculture resource in Mexico, with presumed lack of a negative effect on native biodiversity (Ross and Beveridge 1995). In contrast, Mayan cichlids are one of the most abundant exotic species established in southern Florida (Trexler et al. 2000), where they severely impact native substrate spawners like largemouth bass (*Micropterus salmoides*), warmouth (*Lepomis gulosus*), and spotted sunfish (*L. punctatus*) through nest building, habitat alteration, and egg predation. Since Mayan cichlids outnumber native species in northern Florida Bay, more research is needed on community level impacts in brackish water. Thus, a greater understanding of the life history of the species in low salinity systems in its native range may aid management of introduced populations in south Florida.

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# Gulf and Caribbean Research

Seasonal and Spatial Patterns in Salinity, Nutrients, and Chlorophyll a in the Alvarado Lagoonal System, Veracruz, Mexico

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# SEASONAL AND SPATIAL PATTERNS IN SALINITY, NUTRIENTS, AND CHLOROPHYLL *a* IN THE ALVARADO LAGOONAL SYSTEM, VERACRUZ, MEXICO

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*ABSTRACT* Ten monthly collections, distributed among three seasons, were taken from July 2000 to June 2001 in the Alvarado lagoonal system, Veracruz, Mexico. Variables measured *in situ* included dissolved oxygen, salinity, and water temperature. Water samples were collected to determine concentrations of ammonium, nitrates, nitrites, orthophosphates, total phosphorus and chlorophyll *a*. Collections representing the rainy season were taken in September and October, those for the nortes season were taken in November, December, and January, and dry season collections were taken during February, March, May June, and July. There was seasonal and spatial variation in nutrient concentrations, and they were related to the discharge of the rivers; concentrations increased during the rainy and nortes seasons. Other factors affecting water quality included the constant discharge of organic materials into the system, resuspension of sediments during the nortes season and the biological activity within the system that assimilated the nutrients in the water. The Alvarado lagoonal system has three separate zones based on physicochemical characteristics; Camaronera Lagoon, Buen Pais Lagoon and the urban zone of Alvarado Lagoon, and the river zone in Alvarado Lagoon.

*RESUMEN* Se realizaron diez muestreos durante el periodo comprendido entre Julio de 2000 a Junio de 2001, distribuidos a lo largo de tres estaciones climáticas, en el sistema lagunar de Alvarado, Veracruz, México. Los parámetros que fueron registrados in situ incluyendo oxígeno disuelto, salinidad y temperatura de agua. Al mismo tiempo se colectaron muestras de agua para determinar en laboratorio las concentraciones de amonio, nitratos, nitritos, ortofosfatos, fósforo total y clorofila a. Los meses de colecta que abarcaron las temporada de lluvias fueron tomadas en Julio, Septiembre y Octubre, para Nortes Noviembre, Diciembre, y Enero, y por último, la temporada de secas que correspondieron los muestreos de Febrero, Marzo, Mayo, y Junio. Los nutrientes presentan una variación espacio-temporal presentando relación con: la descarga de los ríos, incrementándose su concentración durante la temporada de lluvias y Nortes; las constantes descargas de agua provenientes de diversas actividades humanas, como son la agricultura y los asentamientos humanos; la resuspensión de los sedimentos durante la temporada de Nortes; y la gran actividad biológica de estaos sistemas que permiten la rápida transformación de la materia orgánica en nutrientes. El sistema lagunar de Alvarado presenta tres zonas diferentes basadas en sus características fisicoquímicas: Laguna de Camaronera, Laguna de Buen País y la zona urbana de Laguna de Alvarado, y la zona de ríos en Laguna de Alvarado.

## INTRODUCTION

Coastal lagoons are productive aquatic systems with a large amount of energetic input. They frequently show elevated concentrations of nutrients (Mee 1978), and many are considered eutrophic. Annually, the constant wind-driven movement of the water column resuspends sediments, which furnishes nutrients to the water column through the biogeochemical cycle and the transformation of materials that were in the sediments (Colombo 1977). Rivers and their drainages provide additional nutrients.

These nutrients can exhibit large seasonal variation, with the highest concentration generally found following a rainy period. Minimal concentrations are detected after the spring phytoplankton bloom, although even in those months, the concentration of nutrients is higher than that in the adjacent coastal zone (De la Lanza and Arenas 1986).

In Mexico, estuarine lagoonal systems represent 30 to 35% of the coastal areas, and 42 of the 134 lagoons are found along the coast of the Gulf of Mexico (GOM) and the Caribbean Sea (Contreras 1985). The estuarine systems along the GOM are generally bordered by well developed

marsh zones, and the ocean influence is accentuated (Kennish 1986). However, in the southern GOM, estuarine systems are generally bordered by mangroves and the degree of oceanic influence varies greatly. In these systems, many factors such as salinity show great seasonal variation. It is common to encounter a gradient where the salinity is higher near the inlets and decreases towards the rivers.

Typically, three seasons (rainy, dry and nortes) define the hydrological behavior of the southern GOM systems (Gómez 1974, Villalobos et al. 1975, Lankford 1977, Botello 1978, Contreras 1988). The rainy season usually occurs from June through September and is characterized by consistent rainfall and large terrestrial runoff, resulting in frequent floods, turbid waters and additional pressures due to the influence of drainage from the land (Contreras 1985). During this season there are brief times of calm weather, characterized by cessation of the rains, high temperatures and elevated rates of evapotranspiration. During these periods, extraordinary photosynthesis occurs, with values occasionally exceeding 700 mgC/m$^3$/h and elevated concentrations of chlorophyll $a$ of 100 mg/m$^3$ (Contreras 1994). The dry season is typically from March through June and has minimal rainfall and river flow (Villalobos et al. 1975, Contreras 1983). The dry season is characterized by elevated temperature, clear water and relative stability in phytoplankton diversity, and the lagoon is generally affected by the dominance of ocean water mass. The higher salinity found during the dry season may be due to evaporation and the reduced influence of the rivers (Villalobos et al. 1966, Contreras 1983). During the nortes, or winter season (October–February), there are strong winds blowing from the north off the GOM and temperatures are low (Herrera and Comin 1995, Barreiro and Aguirre 1999). Autotrophic processes dominate over heterotrophic processes, and there is a considerable quantity of dissolved organic material including organic phosphorus.

The Alvarado lagoonal system is a typical coastal estuary along the southern GOM that supports a variety of different activities, such as fishing, transportation and urban development. A previous study on the hydrography and productivity of this system identified 5 distinct areas, determined by water temperature and chlorinity, within the lagoon: areas with marine influence, areas with freshwater influence, a gradient area, a calm area and the coastal adjacent area (Villalobos et al. 1966). These authors also established that the hydrological and biological characteristics of the lagoon were clearly defined by the rainy and dry seasons. The lagoon is polyhaline with a tendency towards being mesohaline during the dry season and becomes almost totally freshwater during the rainy season (Villalobos et al. 1975). The biological productivity in the

system is high, and primary production and the number of phytoplankton cells are inversely related to the phytoplankton biomass and the postlarval stages of shrimp (Villalobos et al. 1975). More recently, Morán-Silva et al. (1996) reviewed the general hydrological behavior of the Alvarado lagoonal system. They concluded that the hydrological conditions are a direct result of the fluvial discharges and found the lagoon to be predominantly oligohaline. Higher salinity values were found only during the dry season or near the inlets and water temperature varied seasonally. The shallow depth throughout the system, in combination with the winds, allows mixing and aeration of the water column despite the high primary production observed. High nutrient concentrations were found near river mouths, mangroves, and submerged vegetation, presumably through degradation of organic material and resuspension of the sediments. However, the Moran-Silva et al. (1996) study did not examine the seasonal differences in nutrients in the system. Thus, the principal objective of this work is to describe, analyze and characterize the seasonal-scale patterns of the salinity, physicochemical and nutrient variables and their relationship with chlorophyll $a$ during the dry, rainy and nortes seasons in the Alvarado lagoonal system, Veracruz, Mexico.

## Study Area

The Alvarado lagoon system is located in the coastal plain of the GOM, 63 km southeast of the port of Veracruz, between 18°46 and 18°42 N and 95°34 and 95°58 W (Figure 1). Lankford (1977) considered the system to be a drowned river valley. The lagoonal system consists of 3 smaller lagoons with a total length of about 27 km and a surface area of 6,200 ha. Alvarado Lagoon, the main body of water, continues to the west into Buen Pais Lagoon, which is connected to Camaronera Lagoon through a narrow channel to the west. The primary connection to the ocean is Alvarado Inlet, situated at the northeast of the system. A small, 400 m wide outlet to the ocean was constructed in 1982 in Camaronera Lagoon. The Papaloapan River discharges into Alvarado Lagoon from the southeast. Tidal influence does not diminish the outflow of this river, and mean daily flow into the lagoon is 40 million m$^3$ (Contreras 1985). This system is classified as a positive estuary, because the surface water evaporates at a lesser rate than water is added by the river flow (McLusky 1981). This characteristic is in contrast to many lagoonal systems that are hypersaline due to high evaporation and low freshwater input.

The climate of the area is tropical and humid, and precipitation during the summer ranges from 110 to 200 cm. The mean annual temperature varies between 22–26 °C,
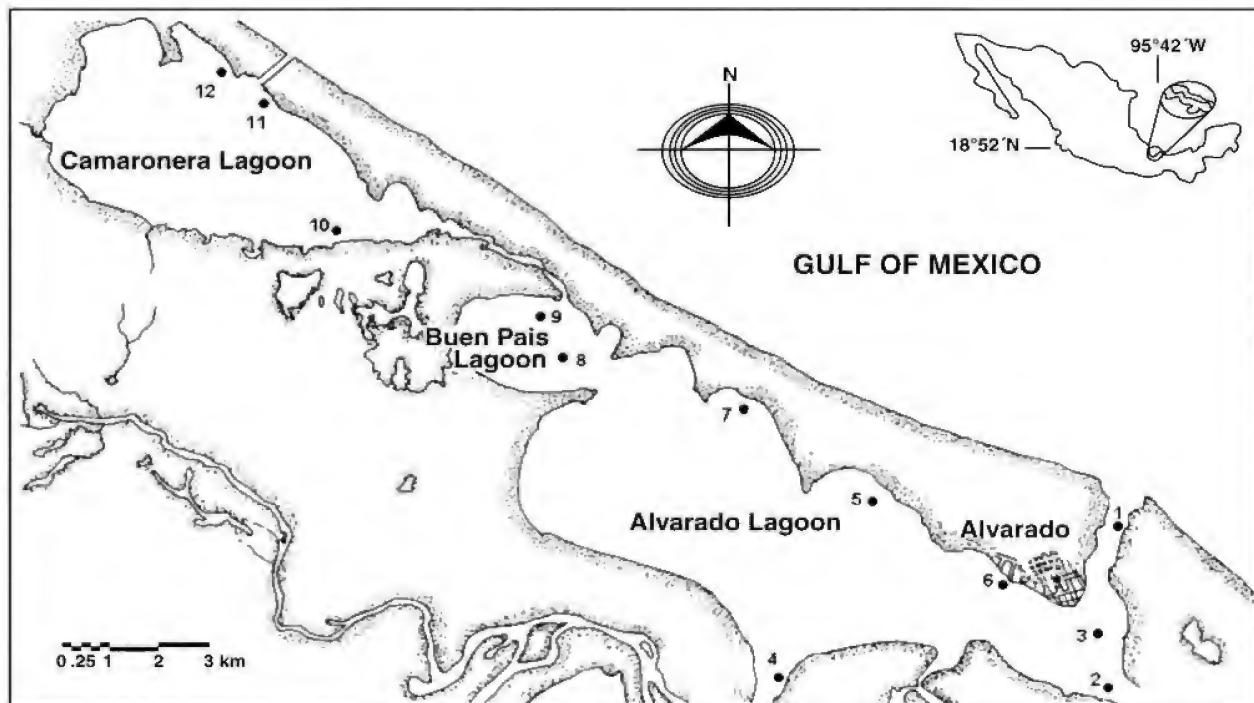
Figure 1. Map of sampling locations in the Alvarado Lagoon system, Mexico.

with temperature oscillations between 5–7 °C between each season (Garcia 1973). The prevailing southeast winds have a maximum velocity of 14.4 msec$^{-1}$, except for October, when winds from the north and northeast range from 90–129.6 msec$^{-1}$ (Contreras 1985).

The lagoon is almost entirely surrounded by mangroves with the typical zonation pattern of red mangrove, *Rhizophora mangle,* bordering the water and black mangrove, *Avicennia germinans*, and white mangrove, *Laguncularia racemosa*, immediately interior. Other sporatically occurring aquatic vegetations include wild celery, *Vallisneria americana*; cordgrass, *Spartina* sp.; and cattail, *Typha* sp.; while the dominant submerged aquatic vegetation (SAV) is *Ruppia maritima* (Morán-Silva et al. 1996). During the rainy season, the water lily, *Eichhornia crassipes*, invades the lagoon.

## MATERIALS AND METHODS

Twelve stations were established throughout the Alvarado lagoonal system (Figure 1) to detect the influence of rivers, inlets, SAV and urban discharges. We defined groups for each system: Camaronera group includes stations 10–12; Buen Pais group includes stations 8–9; the Alvarado Lagoon group (urban dominated) includes stations 5–7; and finally the river dominated group includes stations 1–4 in Alvarado Lagoon. Sampling

occurred at about 30 d intervals between July 2000 and June 2001. Water temperature (°C) was measured with a mercury thermometer, salinity (psu) with a YSI model 33 salinometer and dissolved oxygen (D.O., ml/l) with a YSI model 51b meter during each collection. Surface water samples were collected at each station for nutrient concentration determination. Methods follow Contreras (1994) for ammonium (NH$_4$, mg-at/l), nitrate (NO$_3$, mg-at/l), nitrite (NO$_2$, mg-at/l), phosphate (PO$_4$, mg-at/l), and total phosphorus (P-TOT, mg-at/l). Determination of chlorophyll *a* (chl *a*, μg/l) follows techniques in SCOR-UNESCO (1980). The samples were kept on ice for 48 h prior to analysis (Strickland and Parsons 1972, Wetzel and Likens 1990).

We used correlation analysis to examine the relationship between nutrient concentration and chl *a* (Daniel 1977). Dendrograms were constructed of temporal and spatial classification with Euclidian distance (values range from 0, when entities are identical, to infinity) using the monthly salinity data from each station. Classification of the system using salinity followed the procedure of Carriker (De la Lanza 1994). For all analyses we used the Community Analysis Program (ANACOM) 3.1 (De la Cruz, 1994), and results were considered significant if $P < 0.05$.

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## RESULTS

### Physicochemical variables

Salinity varied among all stations from 14.2 psu during the dry season in June to 0.0 psu during the rainy season in September. The lowest salinity values were always associated with the rivers (0–7.1 psu), whereas the highest values were found in Camaronera and Buen Pais Lagoons (Figure 2a). Salinities at the Alvarado Lagoon stations showed the highest values in March.

Dissolved oxygen varied from 12.8 ml/l in Camaronera Lagoon to 4.13 ml/l in the stations of the river group. The highest D.O. values were found during the dry season, and values for all stations peaked in May (Figure 2b). The D.O. values at the Alvarado Lagoon group stations fluctuated more than those from the other stations.

Water temperature varied seasonally, with annual variation ranging from 21.6 °C during the nortes season to 32.2 °C during the rainy season (Figure 2c). Water temperature tended to be higher in the Alvarado Lagoon zone and lower in the stations with river discharge throughout the year.

### Nutrients

Ammonium was the dominant form of inorganic nitrogen, representing 60.98 to 88.3 % dissolved inorganic nitrogen (DIN). The highest ammonium concentration was 42.43 µg-at/l during the dry season in Buen Pais Lagoon, and the lowest was 2 µg-at/l during the dry season in Camaronera Lagoon (Figure 3a). In general, the highest ammonium values were found at the river group stations and in Buen Pais Lagoon. The highest nitrite concentration was 3.54 µg-at/l during the dry season at the river stations. Undetectable amounts of nitrite were found during the dry season in Buen Pais and during the rainy season in the Alvarado Lagoon group (Figure 3b). Nitrite peaked in the Alvarado Lagoon group (urban zone) at the end of the nortes season. Nitrates were highest during the nortes season in the urban and rivers zones (7.9-10.6 µg-at/l), and lowest (0.67 µg-at/l) in Camaronera Lagoon during the nortes season (Figure 3c). A smaller peak of nitrate was evident in all stations at the end of the dry season.

Camaronera Lagoon had the greatest range in total phosphorus, with highest values during the rainy season (18.8 µg-at/l) and lowest values during the nortes season (3.5 µg-at/l; Figure 4a). Highest values for all stations occurred during the rainy periods (Figure 4a). The values of orthophosphates were highest at the end of the dry season (4.5–6.2 µg-at/l) and lowest during the rainy season (0.37–0.48 µg-at/l) at all stations (Figure 4b).

### Chlorophyll *a*

Chlorophyll *a* values fluctuated during the annual cycle, with lowest values during the dry season (4.3–18.8 µg/l) and highest values in the nortes season (11.5–92.6 µg/l). Buen Pais Lagoon exhibited the greatest fluctuation in chl *a* (Figure 4c). Overall, the river group stations had the lowest chl *a* values (5.1–32.1 µg/l). Correlations between chl *a* and the physicochemical and nutrient measurements differed seasonally, but there were no significant correlations between chl *a* and any other variable measured (Table 1). During the rainy season, there was a moderately positive correlation between chl *a* and salinity and D.O., and a strong negative correlation with ammonium and nitrite. During the nortes season, total phosphorus showed a strong, negative correlation with chl *a*. Salinity had a moderately positive correlation with chl *a* during the dry season, while total phosphorous and phosphates were moderately negatively correlated.

### Spatial-temporal variation

The variability of most of the parameters was reflected principally in salinity, which was rapidly modified by the rain and tidal influence. Using salinity in a cluster analysis, three principal groups were evident (Figure 5a). Group 1 consists of the months of September and October, representing the rainy season, when the system was oligohaline with salinities ranging from 0 to 3.8 psu. Group 2 consists of the months November, December, January and February, corresponding to the nortes season when the salinity begins to increase, ranging from 0 to 11.5 psu. The third group corresponds to the dry season (March, May and June 2001), with the highest salinity values (2 to 14.5 psu), resulting in a mesohaline system. However, July is isolated from the other groups. This month corresponded to the rainy season, while June 2001 was more similar to the dry season due to a lack of rain during that year.

When cluster analysis was applied to the collection stations, the analysis resulted in three groups separated by marine or freshwater influence (Figure 5b). The first group consisted of stations located in Camaronera Lagoon (12 and 11) that receive direct tidal influence through the inlet and had the highest salinities (up to 21 psu). The second group has stations separated into 2 sub-groups, with stations 8–10, representing the eastern portion of Camaronera Lagoon and Buen Pais Lagoon, as one sub-group and stations 5–7, the urban dominated stations, as the second sub-group. The first sub-group receives some tidal influence and had mean salinities ranging from 6.33 to 8.75 psu, while the second sub-group consisted of lower salinity stations located along the eastern shore of Alvarado Lagoon with a marked influence from urban zones. The final major
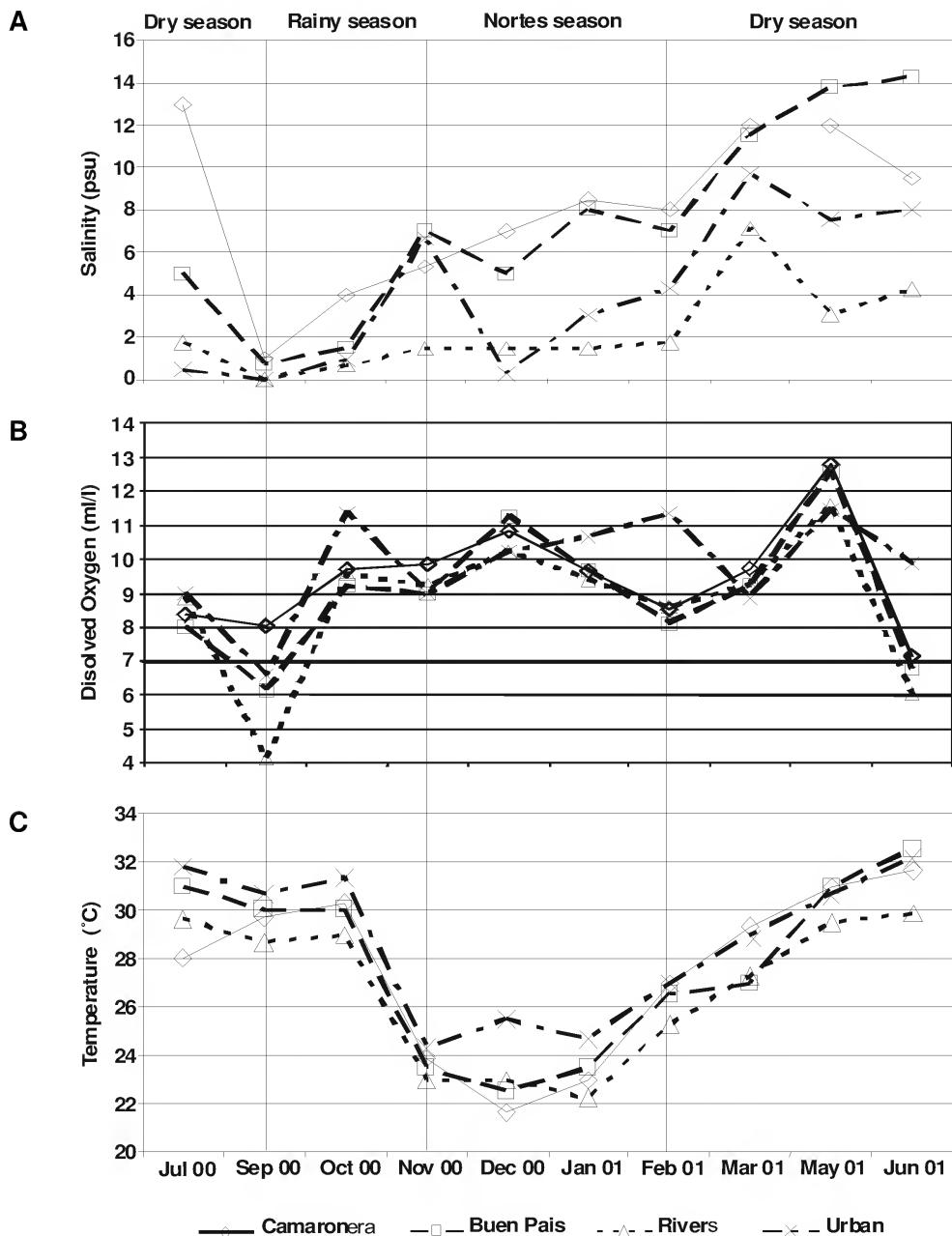
**Figure 2. Plot of mean monthly salinity (A), dissolved oxygen (B), and water temperature (C) over the course of the study pooled by sampling stations within each of the four groups.**

group contained all the stations associated with the river group (1–4) and can be defined as an oligohaline zone.

## DISCUSSION

It is well known that the variability of the hydrological variables and nutrients is especially marked in lagoonal systems. This is due to many factors, like the dynamics in the circulation of the lagoon as affected by the tides, the winds, and the shallow depth. Furthermore, constant resus-

pension of sediments, regeneration processes originated by microbial activity in the sediments, river flow, and human activities contribute to nutrient variation (Colombo 1977, Snedaker and Brown 1982).

As expected, the low salinity values found during the rainy season were a result of the increased freshwater inflow into the lagoonal system (Botello 1978). Similarly, the months corresponding to the dry season (March, May and June in this study) had the highest salinity throughout the system, due to reduced river flow. However, the months
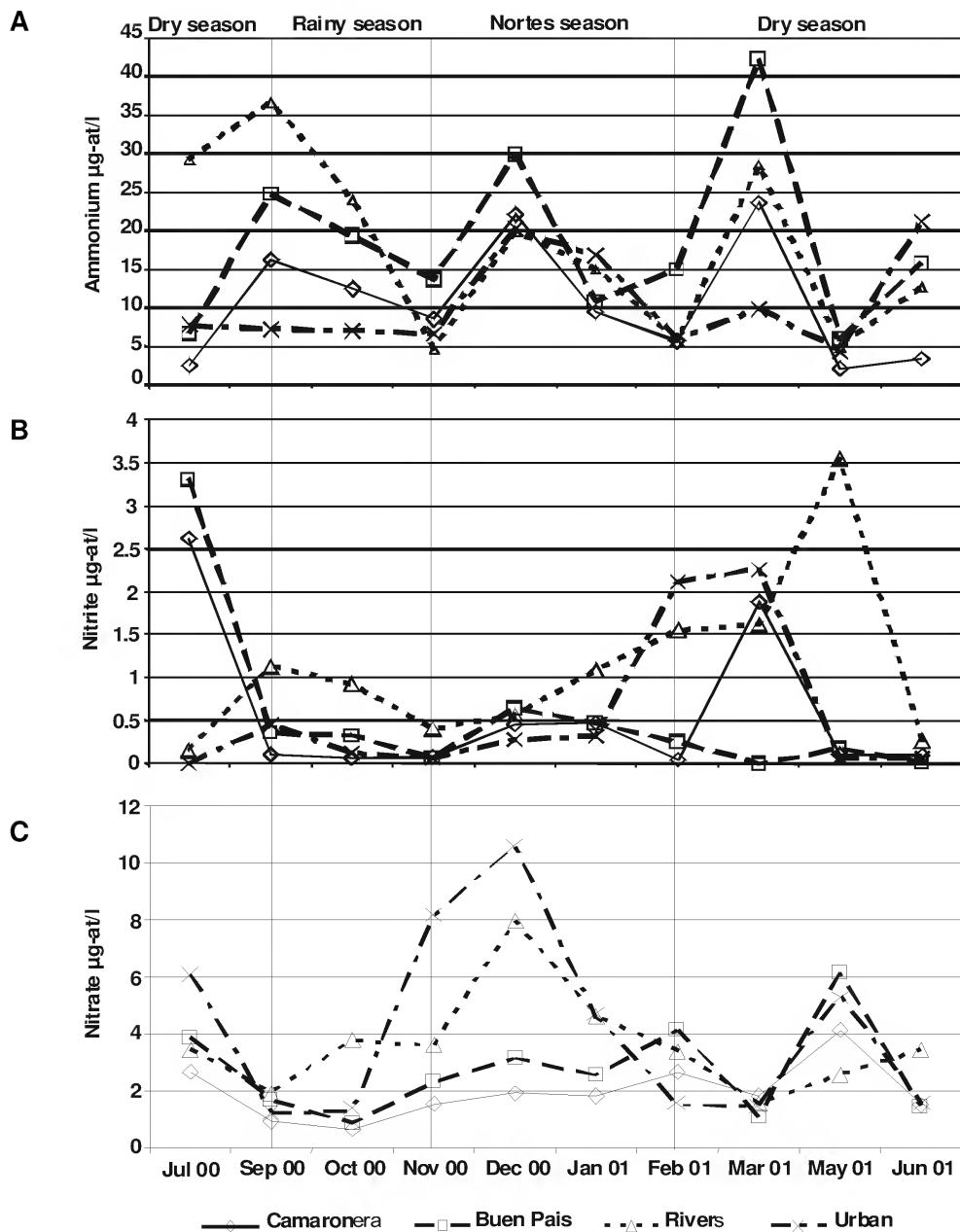
Figure 3. Plot of mean monthly ammonium (A), nitrite (B), and nitrate (C) over the course of the study pooled by sampling stations within each of the four groups.

during the nortes season (December, January and February) had salinity values similar to the dry season. Thus, the Alvarado lagoonal system can be considered oligohaline during the rainy season and mesohaline during the nortes and dry seasons. However, the stations close to the river mouth remained oligohaline during the dry season, indicating a weak marine influence in the lagoon (Morán-Silva et al. 1996). Seasonal differences in salinity have been noted previously in other Mexican lagoons, such as the Celestum Lagoon (Herrera-Silveira and Comin

1995) the Tampamachoco Lagoon (De la Lanza et al. 1998), and the Alvarado lagoonal system (Morán-Silva et al. 1996).

The lowest D.O. concentrations were encountered in September, which corresponds to the end of the rainy season, when there is an increase in suspended organic material. When organic material is resuspended, microorganisms begin decomposition, removing oxygen from the water column (Kennish 1986). The highest D.O. concentrations were found associated with seagrass beds, similar
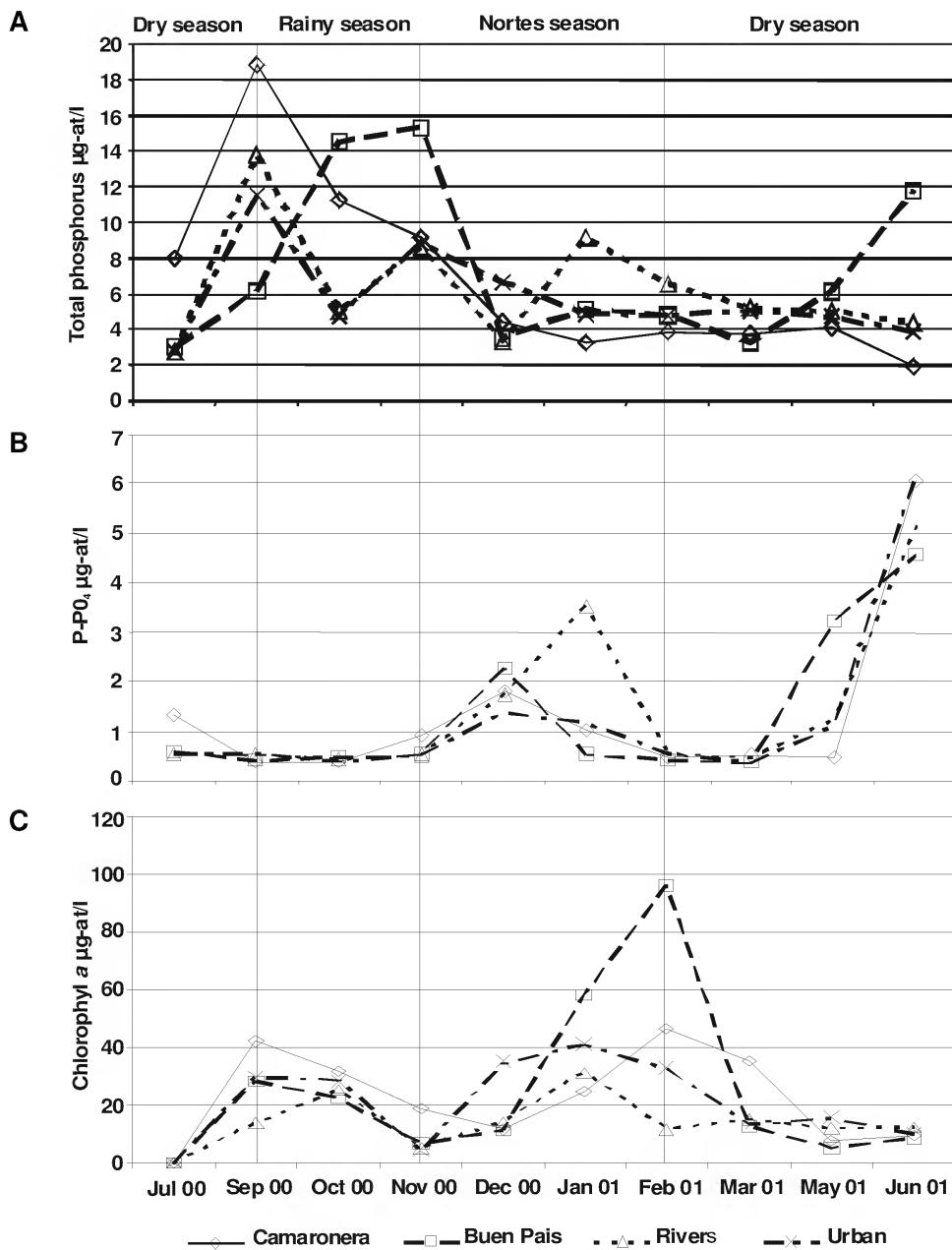
**Figure 4. Plot of mean monthly total phosphorus (A), orthophosphate (B), and chlorophyll *a* (C) over the course of the study pooled by sampling stations within each of the four groups.**

to reports by Contreras and Gutierrez (1989) for other systems in the state of Veracruz. Overall, D.O. concentrations remained relatively heterogeneous and could be related to photosynthetic activity, seasonality, mixing of water and tidal activity (Botello 1978, De la Lanza and Cantu 1986), or to the presence of SAV throughout the system.

With respect to nutrients, the form of ammonia in this type of system comes from degradation of organic material, submerged vegetation and waste from organisms (Tiejten 1968, Botello 1978, De la Lanza and Arenas 1986). Ammonium was the predominant form of inorganic nitrogen during all the seasons. Similarly, Kennish (1986) found that ammonium was the predominant form of inorganic nitrogen in estuarine waters. This agrees with reports by Contreras and Castañeda (1992) and Contreras (1983) for the Tampamachoco Lagoon and the lagoonal system of Carretas-Pereyra, respectively. Maximal ammonium values were observed in Buen Pais Lagoon during the dry season, no doubt enhanced by the increasing temperature that favors a greater degradation of organic mate-
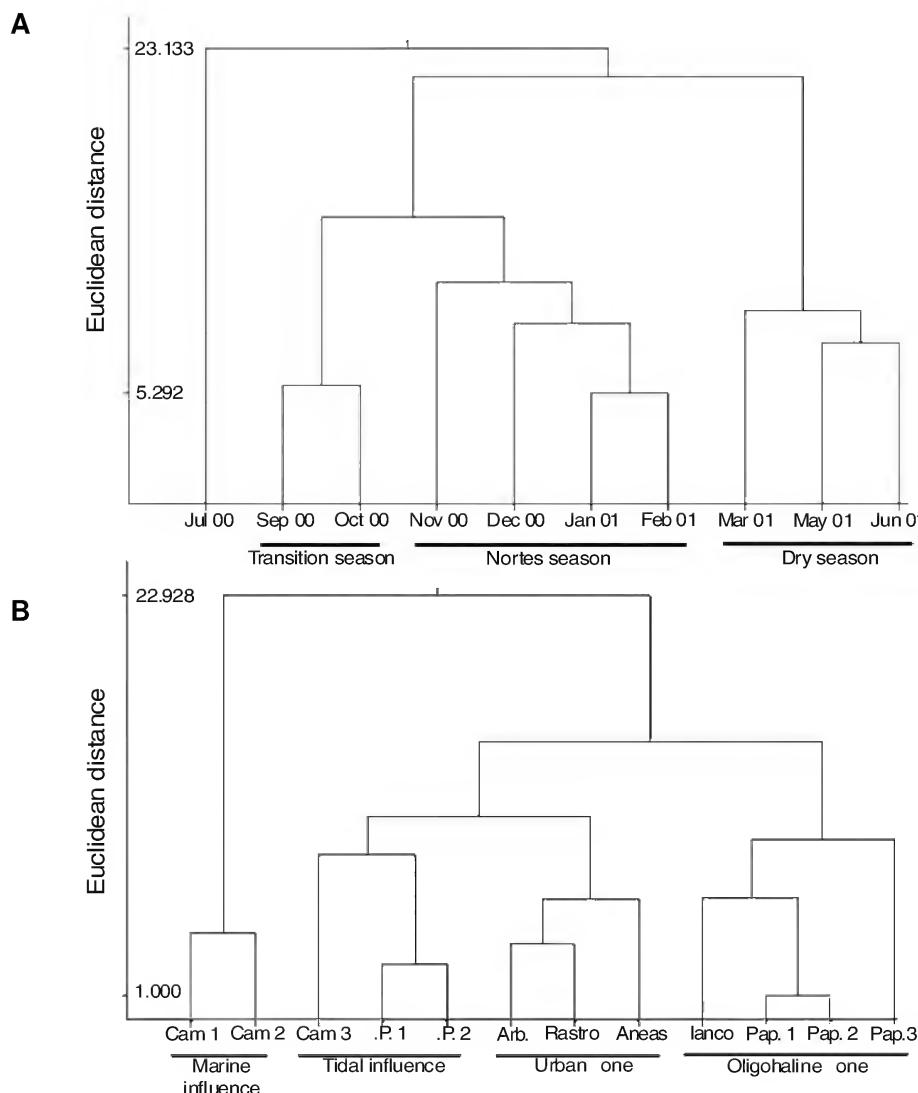
**Figure 5. Dendogram based on Euclidean distance of monthly (A) and station (B) salinity values. Station numbers correspond to stations in Figure 1. Cam-Camaronera Lagoon; B.P.-Buen Pais Lagoon; Arb., Rastro, Aneas—urban areas on the north side of Alvarado Lagoon; Blanco-Blanco River, Alvarado Lagoon; Pap.-Papaloapan River, Alvarado Lagoon**

rials found in the sediments as well as increased waste from organisms in the water column. However, Kennish (1986) indicated that the concentrations of the nitrogenous components can be augmented with the river flow. We observed a similar increase in the river group stations during September, a time of high river flow in Alvarado Lagoon. Day et al. (1998) reported a similar situation in Terminos Lagoon, showing augmentation of nutrient concentrations during times of high discharge from the rivers.

Buen Pais Lagoon had generally higher values of ammonium than the other two lagoons within the system, with a peak in March. This may be due to slower water circulation in this lagoon relative to the others in the system (Villalobos et al. 1975). In contrast, the urban areas of

Alvarado Lagoon did not have an increase in nitrogen in March, although there were peaks in June and December. Alvarado Lagoon is impacted by urban discharges from the Port of Alvarado, which tend to increase the nitrogen concentration in the water (Barreiro and Aguirre 1999). This was particularly evident for nitrate during the nortes season.

The Lagoon showed notable hydrological variation on spatial as well as temporal scales. For instance, eutrophication was noted in semi-isolated areas such as within canals, which had minimal effects of circulation, yet the rest of the Lagoon was not eutrophic. On a temporal scale, the dry and wet seasons result in changes in salinity and nutrients with consequent variation in the habitat during the annual cycle.

**TABLE 1**

**Seasonal correlation coefficients (Pearson's r) between chlorophyll *a* and various water chemistry variables in the Alvarado lagoonal system. No values were significant at *P* < 0.05.**

| Variables | Rainy | Nortes | Dry |
|---|---|---|---|
| Salinity (psu) | 0.55 | −0.04 | 0.43 |
| Dissolved oxygen (ml/l) | 0.59 | −0.15 | 0.02 |
| Temperature (°C) | 0.35 | −0.24 | 0.29 |
| Ammonium (μg-at/l) | −0.65 | 0.29 | 0.10 |
| Nitrite (μg-at/l) | −0.14 | 0.27 | −0.20 |
| Nitrate (μg-at/l) | −0.74 | −0.13 | 0.15 |
| Phosphate (μg-at/l) | −0.15 | 0.14 | −0.37 |
| Total phosphorous (μg-at/l) | −0.20 | −0.62 | −0.47 |

The nutrient concentrations reached during the rainy season were more elevated than during the dry season.

The Alvarado lagoonal system had the highest phosphorus concentration during the rainy season. While this nutrient comes principally from organic material, it is also produced through autochthonous processes such as bioturbation and remineralization of the sediments and remixing by currents (Groen 1969). Total phosphorus was highest during September throughout the lagoonal system, no doubt due to the effects of increased river runoff and resuspension of the sediments (Groen 1969). Concentration decreased gradually to the lowest point during the dry season. A peak of orthophosphates in June may be related to the decrease in chl *a* concentration during this month, as phytoplankton utilize orthophosphates (Contreras and Castaneda 1992). Overall, Camaronera Lagoon had the lowest concentration of phosphates, probably because currents are minimal, resulting in little resuspension of the sediment where the majority of phosphates are stored (De la Lanza 1996). In Alvarado Lagoon, phosphates were higher during the nortes and dry seasons compared to the rainy season, and variation was not as great as in the other lagoons. The variation in phosphate that was observed is probably a direct result of river input. The agricultural land and associated fertilizers within the drainage basin of the Papaloapan River are important sources of phosphates (Correll et al. 1992), which can be transported into the lagoon through erosion and runoff.

A global characteristic of lagoonal phytoplankton is their high productivity. For this reason, we consider coastal lagoons as ecosystems with characteristics intermediate between the ocean and the rivers (Margalef 1969). Since algae are the only organisms that remain constant with respect to other cellular components that are ecologically

important, chl *a* concentration can be used to better understand the dynamics of the system (Marshall 1987). Unfortunately, the coefficients of correlation did not show a significant relation between the concentration of chl *a* and the physicochemical variables. There is usually a strong relationship between nutrient concentration and chl *a* concentration, as has been previously discussed (Contreras 1994). We found an increase from undetectable chl *a* in July to moderate levels (14–42 μg/l) in September and October, similar to findings in other Mexican lagoonal systems (Contreras et al. 1992, Contreras and Castaneda 1992, Barriero and Aguirre 1999). Interestingly, the highest correlation between D.O. and chl *a* was found during this time, when chl *a* began to increase from a dry season low, suggesting an increase in productivity. During November and December, chl *a* again decreased in most areas of the lagoon, and chl *a* values were higher from January through March, with a peak in February. The high chl *a* values in Buen Pias Lagoon during February (96.2 μg/l) indicate a hypereutrophic system at that time. The February peak corresponds to the end of the nortes season, a time when Li et al. (2000) found an association of phytoplankton blooms with a peak of nutrients. The decrease in chl *a* during May and June may be related to the increase in phosphates and inorganic phosphorus during this time.

Barreiro and Aguirre (1999) found that an increase in nitrate during the rainy season is necessary for a phytoplankton bloom to commence during the dry season. Our data show a dramatic increase in nitrate during November and December, which may be related to the bloom, and a subsequent increase in chl *a* in February. The predominance of blooms during the dry season may also be related to calmer water conditions during this time (Marshall 1987).

Overall, the pattern of chl *a* was relatively similar among stations and lagoons, with peaks and low points occurring during similar times. Spatial patterns of chl *a* respond to local conditions (Barreiro and Aguirre (1999), and the estuarine currents can distribute the phytoplankton biomass asymmetrically (Li et al. 2000). For instance, phytoplankton populations from the ocean may enter the lagoon on incoming tides (Revilla et al. 2000), which may explain the increased chl *a* concentration near the inlet in Camaronera Lagoon during September, January and February. On the other hand, Revilla et al. (2000) found that the major concentration of chl *a* in estuaries was found in discharge areas that did not receive a direct tidal influence. Similarly, Day et al. (1998) found a major concentration of chl *a* in Estero Pargo (mean annual value 8 μg/l) in comparison to Terminos Lagoon (3 μg/l). However, our

data show that the chl *a* concentration was low at stations located at the Papaloapan and Blanco rivers, where there is major discharge but minor tidal influence.

In terms of spatial distribution, it is possible to distinguish areas directly influenced by terrestrial sources due to elevated quantities of phosphorus. These are interpreted as areas within the lagoonal system with a greater density of primary producers, compared to other zones where different factors, such as circulation, river influence, or winds do not permit the accumulation of phytoplankton. The persistence of these phytoplankton overloaded areas is the direct cause of natural eutrophication or eutrophication originated by urban activities. Natural eutrophication is a result of geographic properties, accumulation of sediment, etc., while anthropogenic eutrophication is a result of uncontrolled use of fertilizers, deforestation, and addition of contaminants and human wastes to the lagoonal system. The continued urban development along the internal coast of the Alvarado Lagoon exacerbates the anthropogenic inputs to the system. Finally, there has been a change in the bottom use in the discharge area of the Papaloapan River that has altered the hydrological dynamics.

Our results suggest that habitats within the lagoonal system have high heterogeneity that is driven by variation in salinity and water temperature. This variation is the result of the influence of river discharge and tidal exchange. In addition, these difference may also relate to the bathymetry, the presence of SAV or the proximity of mangroves. However, our results do not correspond to those reported by other authors. Villalobos et al. (1966) described 5 natural areas based on the influence of the rivers and the ocean, whereas in this work, we define only 3 such areas, which are a function of river discharge, proximity to ocean inlets and the influence of urban discharges. Our findings concur with Lozano (1993) who found that an increase of anthropogenic activities, in conjunction with poor planning, contributed to local and regional changes in hydrological characteristics of the freshwater sources to the Alvarado lagoonal system.

Salinity characteristics of the Alvarado lagoonal system vary seasonally. Our work has reinforced the observations of Villalobos et al. (1975) who described the seasonal salinity variation. Furthermore, two earlier studies on the Alvarado lagoonal system found that salinity varied more than other variables (Sevilla and Chee 1974) and was lowest during the rainy period (Sevilla and Chee 1974, Chee 1981). It appears that the amount of rainfall and subsequent river discharge is one of the forces driving the variability of the system. Thus, to better understand the hydrology of the Alvarado lagoonal system this information is required.

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# LONG-TERM FISH ASSEMBLAGE DYNAMICS OF THE ALVARADO LAGOON ESTUARY, VERACRUZ, MEXICO

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*ABSTRACT* The fish assemblages of Alvarado Lagoon Estuary (a complex of coastal lagoons in the state of Veracruz, Mexico) have been surveyed intermittently by different researchers over the last 40 years. Assessing long-term trends in fish assemblage composition for this ecosystem is problematic due to differences in sampling efforts among the survey periods (1966–1968, 1987–1988, 1989, 1989–1990, 1990–1991 and 2000–2001) and by the inherent ecological variability of estuaries. To overcome these data limitations and better understand fish assemblage change over time, we used robust, simulation-based analyses to compare collections from the different surveys. The 107 fish species collected from the Alvarado Lagoon Estuary in these surveys represent 4 ecological guilds: marine stenohaline, marine euryhaline, estuarine, and freshwater fishes. The occurrence frequency of fish species representing each guild did not change significantly among the survey periods: the chi-square deviation statistic ( $x^2$ = 8.53) was not significantly larger than the average value for 1000 simulated matrices ( $x^2$ = 138.64; $P$ = 1.00). A non-metric multidimensional scaling (MDS) based on Bray-Curtis similarities of fish species presence-absence data showed that the 1966–1968 survey period was the least similar to the other survey periods. For the 1966–1968 survey, the range of Bray-Curtis inter-survey similarities was 40.4–58.6 ($n$ = 5). By comparison, the remaining range of inter-survey similarities was 61.5–81.7 ($n$ = 10). Average taxonomic distinctness ($\Delta^+$) and variation in taxonomic distinctness ($\Lambda^+$), two sample size-independent measures of diversity, were calculated for all survey periods. Although $\Delta^+$ and $\Lambda^+$ for all survey periods were within the simulated 95% confidence limits for expected values, these values for the 2000–2001 survey period were less than the average $\Delta^+$ and $\Lambda^+$ values for the entire species pool. This suggests that the fish assemblage collected during the latest survey reflects a loss of both widespread higher taxa (reduced $\Delta^+$) and that the higher taxa lost are those with only a few representative species in the assemblage (reduced $\Lambda^+$). These assemblage data show that fish assemblages of Alvarado Lagoon Estuary have not experienced significant changes over 40 years, but differences among the earliest (1966–1968), the latest (2000–2001), and the remaining survey periods indicate a recent decline in diversity.

*RESUMEN* Los ensamblajes de peces del sistema lagunar de Alvarado (un complejo de lagunas costeras del Estado de Veracruz, México) han sido investigados intermitentemente por diferentes grupos durante los últimos 40 años. La determinación de las tendencias a largo plazo de estos ensamblajes ha sido problemática debido a las diferencias en los esfuerzos de muestreo empleados entre los períodos de investigación (1966–1968, 1987–1988, 1989, 1989–1990, 1990–1991 y 2000–2001) y por la variación ecológica inherente a los estuarios. Para evitar estas limitaciones de los datos y presentar una explicación apropiada de los cambios de los ensamblajes de peces respecto al tiempo, se usó un análisis de simulación para comparar las colectas de los diferentes períodos de trabajo. Las 107 especies colectadas en el sistema lagunar de Alvarado representan 4 gremios ecológicos: marino estenohalino, marino eurihalino, estuarino y dulceacuícolas. La frecuencia de ocurrencia de las especies de peces que representan cada gremio no cambió significativamente entre los períodos de investigación: El estadístico de desviación chi cuadrada ( $x^2$ = 8.53) no fue significativamente mas grande que el valor promedio para las 1000 matrices simuladas ( $x^2$ = 138.64; $P$ = 1.00). La prueba de escalamiento múltiple dimensional no-métrico (MDS) se ejecutó considerando los datos de presencia-ausencia y las similitudes calculadas por el índice de Bray-Curtis, esta prueba mostró que el período 1966–1968 fue menos similar a los otros períodos de colecta. Para el período 1966–1968, el rango de similitudes Bray-Curtis entre investigaciones fue de 40.4–58.6 ($n$ = 5). En comparación, el rango de similitudes restante entre investigaciones fue de 61.5–80.7 ($n$ = 10). La distinción taxonómica promedio ($\Delta^+$) y la variación de la distinción taxonómica ($\Lambda^+$), dos medidas de la diversidad independientes del tamaño de muestra, fueron calculadas para todos los períodos de investigación. Aunque los valores de $\Delta^+$ y $\Lambda^+$ para todos los períodos de investigación estuvieron dentro de los límites de confianza de 95% para los valores esperados, estos valores fueron menores para el período 2000–2001 respecto a los valores promedio de $\Delta^+$ y $\Lambda^+$ para el conjunto completo de especies. Esto sugiere que el ensamblaje de especies colectado en el último período de investigación refleja una pérdida de taxa superiores ($\Delta^+$ reducida) y que los taxa superiores perdidos son aquellos con pocas especies representativas en el ensamblaje ($\Lambda^+$ reducida). Estos datos muestran que los ensamblajes de especies del sistema lagunar de Alvarado no han sufrido cambios significativos durante los últimos 40 años, pero las diferencias entre el primer y último período de investigación y los otros períodos indican una declinación reciente en la diversidad.

## INTRODUCTION

Coastal lagoons are interface ecosystems that have a rich biodiversity of organisms due to their position between river drainages and the continental shelf. The environmental gradients within coastal lagoons typically allow for numerous species with various life cycles, resulting in the formation of communities whose structures are influenced by physical and chemical factors. These ecosystems are dynamic and vary seasonally, which influences the presence of organisms from marine or freshwater origin and creates an environment that is critical for commercially important species such as molluscs, crustaceans, and fishes (Beck et al. 2001, Blaber 2002).

The Alvarado Lagoon Estuary is a complex of coastal lagoons located in southern Veracruz, Mexico. The Papaloapan, Blanco, and Acula rivers drain into the estuary, which in turn drains north into the Gulf of Mexico near Alvarado Port. The largest of these river basins, the Papaloapan River Basin, is more than 500 km long, covers an area of 46,517 km², and has an annual discharge of 47 million m³. The combination of numerous freshwater sources and multiple connected lagoon complexes forms a rich coastal ecosystem in the Alvarado Lagoon Estuary (Chávez-Lopez 1998). However, environmental and ecological changes due to anthropogenic factors such as overexploitation of the resources, industrial contamination, and construction of dams have contributed to the progressive decrease in the quality and ecological value of estuaries (Whitfield and Elliot 2002). This increasing alteration of environmental quality and quantity contrasts with the recent classification of lagoonal systems as strategic areas for the conservation of biodiversity (CONABIO 1998, Zárate-Lomelí et al. 1999). If the biodiversity of the Alvarado Lagoon Estuary is to be protected, it is necessary to determine which organisms are consistently present in the ecosystem and which appear to be declining in response to increased anthropogenic impacts.

Although the fish assemblages of the Alvarado Lagoon Estuary have been surveyed intermittently over the last 40 years by various researchers, these data have never been analyzed as an entirety to assess possible changes in fish biodiversity over that period. We compared fish collection data for 6 surveys (1966–1968, 1987–1988, 1989, 1989–1990, 1990–1991 and 2000–2001) to determine if the fish assemblages had changed among the survey periods. More specifically we addressed the following questions: 1) Did the frequency of marine stenohaline, marine euryhaline, estuarine, and freshwater fishes change significantly among surveys?, 2) Did assemblages remain similar throughout the 40 year period as determined by Bray-Curtis similarity indices?, and 3) Did biodiversity decline significantly over the same period as determined by comparisons of average taxonomic distinctness and variation in taxonomic distinctness (two sample size-independent measures of biodiversity)?

## Study Area

The Alvarado Lagoon Estuary is comprised of the Alvarado, Buen Pais, and Camaronera lagoons and the estuarine zone of the Papaloapan River (Figure 1). This estuarine system has a total surface area of 6,200 ha, with a mean depth of 2.5 m in the central zone of the lagoon and 14 m in the channel of the Papaloapan River. Water temperatures from April to September range between 27–33 °C, while during winter months (December to February) water temperatures reach as low as 22 °C. The salinity varies from 0–10 psu between July and October (rainy season), although during this period salinities are higher at the artificial inlet in Camaronera Lagoon. From November through June (nortes and dry seasons) salinities average 16 psu in Camaronera Lagoon and 22 psu at the mouth of the Papaloapan River, whereas the rest of Alvarado and Buen Pais lagoons have salinities ranging from 0–8 psu.

## MATERIALS AND METHODS

We compiled fish assemblage data for the Alvarado Lagoon Estuary from previous research reports that described past fish surveys (Reséndez 1973, Chávez-López 1998). Reséndez (1973) provided a list of species collected between 1966 and 1968 using various fishing gear but did not provide precise information on capture methods, abundance, biomass, or physico-chemical variables. Chávez-López (1998) included a list of species as well as abundance, biomass, and ecological variables of the assemblage from November 1987 to August 1991. Our own monthly collections from June 2000 to August 2001 provided the latest data from the Estuary. We sampled fishes at 12 stations which included various habitat types: submersed aquatic vegetation (*Ruppia maritima*) (stations 2, 3, 4, and 6), old oyster reefs (station 5), river mouths (stations 7 and 10), the estuarine zone of the Papaloapan River (stations 11 and 12), a station near the artificial inlet that did not have submersed aquatic vegetation (station 1), and 2 stations located near Alvarado Port with urban influence (stations 8 and 9; Figure 1). Our fish sampling method consisted of a single seine haul with a 30 m long x 2 m high seine with 19 mm mesh, resulting in an effective collection area of 27.5 x 27.5 m (756.2 m²). Seines were operated perpendicular to shore at all stations except 11, where
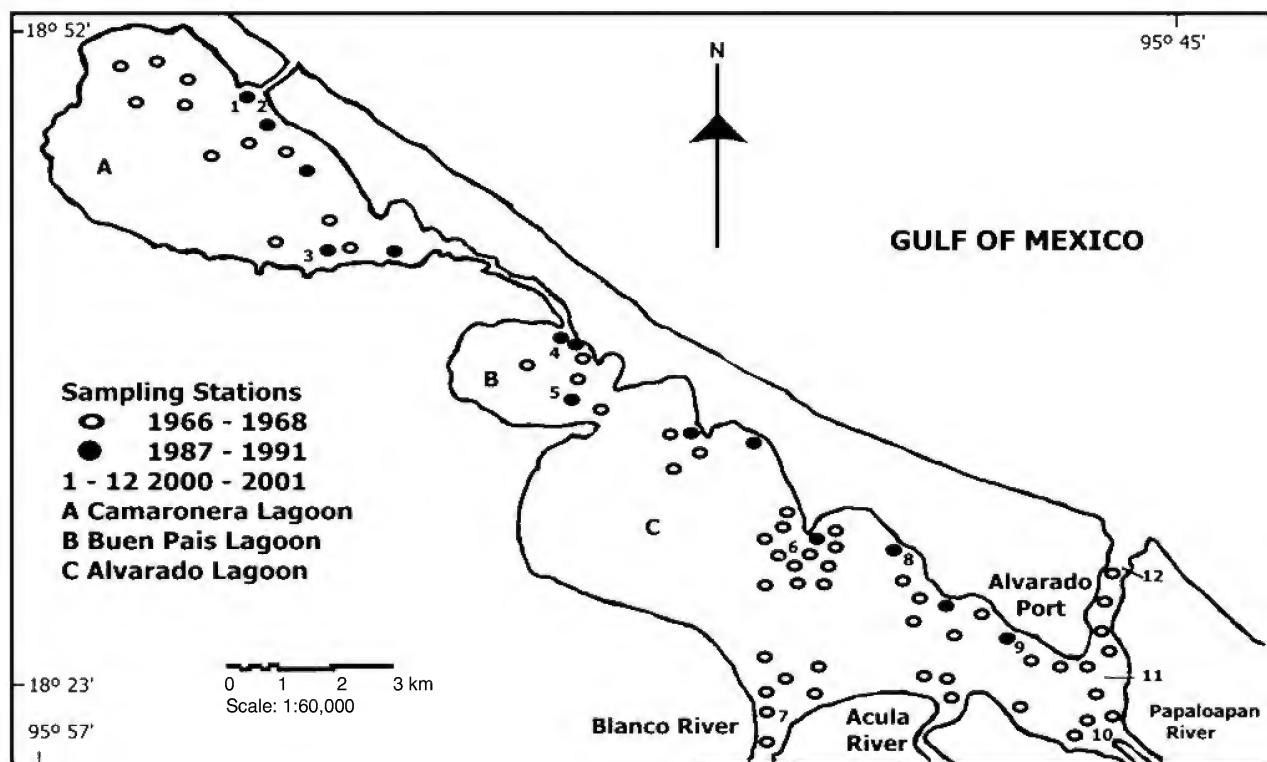
**Figure 1.** Map of the Alvarado Lagoon Estuary showing sampling stations for 6 fish surveys conducted during the following periods: 1966–1968, 1987–1988, 1989, 1989–1990, 1990–1991 and 2000–2001.

the same area was sampled in a circular pattern. The mean depth of all stations was 0.98 m. All organisms were fixed in 10% formalin and later washed in tap water and preserved in 70% ethanol. Species identification and common names were verified based on Reséndez (1973, 1981, 1983), Fischer (1978), Arredondo and Guzmán (1987), Hubbs et al. (1991), Hoese and Moore (1998), and Nelson et al. (2004).

From the lists of fishes, we constructed a presence-absence matrix of species based on the 6 collection periods: 1966-1968, November 1987–September 1988, February–September 1989, November 1989 to June 1990, November 1990 to August 1991, and June 2000 to June 2001 (Table 1). All seasons are represented in each collection period except 1989 when no collections were made during the nortes season (November through February). We classified fishes of the Alvarado Lagoon Estuary into 4 ecological guilds: marine stenohaline (M), marine euryhaline (ME), estuarine (E), and freshwater (F). These guilds were based on descriptions in Castro-Aguirre et al. (1978), Chávez-Lopez (1998), and Ross et al. (2001). If a fish specimen could not be identified to species level, it was recognized as "sp." under its genus epithet and included in the list as a separate species. We used EcoSim software (v. 7.58) to determine if the occurrence frequency of marine

stenohaline, marine euryhaline, estuarine, and freshwater fishes differed significantly among surveys. A matrix representing the number of species from each guild collected from each survey was compared to a matrix representing expected values. For this comparison, expected frequencies were represented as the mean frequencies of each guild across all surveys. A chi-square statistic was generated to quantify the level of deviation between the observed and expected matrices. Once this statistic was calculated, EcoSim was used to create 1000 simulated matrices based on a randomization of the observed matrix. For this case, randomization of frequencies was designated to operate across surveys but not across guilds. In other words, if Guild A consistently had roughly twice as many species as Guild B for all 6 surveys, then all simulated matrices reflected this reality. A chi-square deviation statistic was calculated for the comparison of each of these 1000 simulated matrices to the expected mean frequencies. The resulting 1000 chi-square statistics were plotted as a frequency diagram and compared with the original observed-expected chi-square statistic. If the observed frequencies of species in guilds deviated significantly from random, the original observed-expected chi-square statistic should be greater than at least 950 chi-square statistics generated by the simulated matrices ($P = 0.05$).

## TABLE 1

**Fish species collected from the Alvarado Lagoon Estuary during 6 separate surveys from 1966 to 2001. Each species was assigned to one of 4 ecological guilds: marine stenohaline (MS), marine euryhaline (ME), estuarine (E), or freshwater (F). Species occurrence in a collection is denoted by an X with non-occurrences denoted by an 0.**

| Species | Ecological Guild | 1966–1968 | 1987–1988 | 1989 | 1989–1990 | 1990–1991 | 2000–2001 |
|---|---|---|---|---|---|---|---|
| *Dasyatis sabina* | ME | X | X | X | X | X | X |
| *Elops saurus* | ME | X | X | 0 | X | X | X |
| *Myrophis punctatus* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Harengula jaguana* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Opisthonema oglinum* | ME | 0 | X | X | X | X | X |
| *Brevoortia gunteri* | ME | X | 0 | 0 | X | 0 | 0 |
| *Brevoortia patronus* | ME | 0 | 0 | 0 | X | 0 | 0 |
| *Dorosoma cepedianum* | F | 0 | X | X | X | X | 0 |
| *Dorosoma petenense* | F | X | 0 | X | X | X | 0 |
| *Anchoa hepsetus* | MS | 0 | X | 0 | 0 | 0 | 0 |
| *Anchoa mitchilli* | ME | X | 0 | X | X | X | X |
| *Cetengraulis edentulus* | MS | 0 | X | 0 | 0 | X | 0 |
| *Synodus foetens* | MS | 0 | 0 | 0 | X | 0 | 0 |
| *Astyanax fasciatus* | F | 0 | 0 | 0 | 0 | 0 | X |
| *Ictiobus meridionalis* | F | X | 0 | 0 | 0 | 0 | 0 |
| *Arius felis* | ME | X | X | X | X | X | X |
| *Cathoropus melanopus* | E | X | X | X | X | X | X |
| *Bagre* sp. | ME | 0 | 0 | 0 | X | 0 | 0 |
| *Bagre marinus* | ME | X | X | 0 | X | X | X |
| *Rhamdia guatemalensis* | F | 0 | 0 | 0 | 0 | 0 | X |
| *Opsanus beta* | ME | X | X | 0 | X | X | X |
| *Gobiesox strumosus* | ME | X | 0 | 0 | 0 | 0 | 0 |
| *Hemirhamphus brasiliensis* | ME | 0 | 0 | X | 0 | 0 | 0 |
| *Hyporhamphus roberti* | ME | 0 | X | X | X | X | X |
| *Strongylura marina* | ME | X | X | X | X | X | X |
| *Strongylura notata* | ME | 0 | X | X | X | X | X |
| *Strongylura timucu* | MS | 0 | 0 | 0 | 0 | 0 | X |
| *Poecilia mexicana* | F | X | X | X | X | X | X |
| *Belonesox belizanus* | F | X | 0 | 0 | 0 | 0 | X |
| *Menidia beryllina* | ME | 0 | 0 | 0 | 0 | X | X |
| *Membras vagrans* | MS | 0 | X | 0 | 0 | 0 | 0 |
| *Thyrinops* sp. | F | X | 0 | 0 | 0 | 0 | 0 |
| *Syngnathus louisianae* | ME | 0 | X | 0 | 0 | X | 0 |
| *Syngnathus scovelli* | ME | X | 0 | 0 | 0 | X | X |
| *Microphis brachyurus* | ME | X | 0 | 0 | 0 | 0 | X |
| *Ophisternon aenigmaticum* | F | X | 0 | 0 | 0 | 0 | 0 |
| *Prionotus punctatus* | MS | 0 | X | 0 | X | X | 0 |
| *Centropomus undecimalis* | ME | X | X | 0 | X | X | X |
| *Centropomus parallelus* | ME | X | X | X | X | X | X |
| *Centropomus poeyi* | ME | X | 0 | 0 | 0 | 0 | 0 |
| *Centropomus ensiferus* | ME | 0 | 0 | X | 0 | 0 | X |
| *Centropomus pectinatus* | ME | 0 | X | X | X | X | X |
| *Caranx latus* | MS | X | 0 | 0 | X | X | X |

**Table 1 (continued)**

| Species | Ecological Guild | 1966–1968 | 1987–1988 | 1989 | 1989–1990 | 1990–1991 | 2000–2001 |
|---|---|---|---|---|---|---|---|
| *Caranx hippos* | MS | X | X | 0 | 0 | X | X |
| *Caranx crysos* | MS | 0 | 0 | 0 | 0 | 0 | X |
| *Selene vomer* | MS | X | 0 | X | X | X | X |
| *Caranx bartholomei* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Hemicaranx amblyrhynchus* | MS | 0 | X | X | 0 | X | 0 |
| *Trachinotus carolinus* | ME | 0 | X | X | X | X | 0 |
| *Trachinotus falcatus* | ME | 0 | 0 | X | X | X | X |
| *Oligoplites saurus* | ME | 0 | X | X | X | X | X |
| *Lutjanus synagris* | MS | 0 | 0 | 0 | 0 | 0 | X |
| *Lutjanus apodus* | MS | X | X | 0 | 0 | 0 | 0 |
| *Lutjanus griseus* | MS | X | X | X | X | X | 0 |
| *Lutjanus jocu* | MS | 0 | X | 0 | X | X | 0 |
| *Eucinostomus gula* | MS | 0 | X | 0 | 0 | 0 | 0 |
| *Eucinostomus melanopterus* | ME | X | X | X | X | X | X |
| *Diapterus rhombeus* | ME | X | X | X | X | X | X |
| *Diapterus auratus* | ME | X | X | X | X | X | X |
| *Gerres cinereus* | MS | 0 | X | X | 0 | 0 | 0 |
| *Eugerres plumieri* | MS | X | X | X | X | X | X |
| *Haemulon plumieri* | MS | 0 | 0 | 0 | 0 | X | 0 |
| *Conodon nobilis* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Pomadasys croco* | MS | X | 0 | 0 | 0 | X | X |
| *Archosargus rhomboidalis* | MS | 0 | X | 0 | X | X | 0 |
| *Archosargus probathocephalus* | ME | X | X | 0 | X | X | 0 |
| *Lagodon rhomboides* | MS | X | X | 0 | X | X | 0 |
| *Cynoscion nothus* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Cynoscion nebulosus* | MS | X | 0 | 0 | 0 | 0 | 0 |
| *Bairdiella ronchus* | ME | X | 0 | 0 | X | X | 0 |
| *Bairdiella chrysoura* | ME | X | X | X | X | X | X |
| *Stellifer lanceolatus* | ME | 0 | X | X | X | X | X |
| *Micropogonias furnieri* | ME | X | X | X | X | X | X |
| *Chaetodipterus faber* | MS | 0 | 0 | 0 | X | 0 | 0 |
| *Cichlasoma octofasciatum* | F | X | 0 | 0 | 0 | 0 | 0 |
| *Cichlasoma salvini* | F | 0 | 0 | 0 | 0 | 0 | X |
| *Vieja fenestrata* | F | X | 0 | 0 | 0 | 0 | 0 |
| *Cichlasoma urophthalmus* | F | 0 | X | X | X | X | X |
| *Cichlasoma synspillum* | F | 0 | 0 | 0 | X | 0 | X |
| *Cichlasoma champotonis* | F | 0 | 0 | 0 | 0 | 0 | X |
| *Cichlasoma helleri* | F | 0 | X | X | X | X | 0 |
| *Cichlasoma* sp. | F | 0 | 0 | 0 | 0 | 0 | X |
| *Petenia splendida* | F | 0 | X | X | X | X | X |
| *Oreochromis aureus* | F | 0 | X | X | X | X | X |
| *Oreochromis niloticus* | F | 0 | X | X | X | X | X |
| *Mugil curema* | ME | X | X | X | X | X | X |
| *Mugil cephalus* | ME | X | X | X | X | 0 | X |
| *Mugil gaimardianus* | MS | 0 | 0 | X | 0 | 0 | 0 |
| *Agonostomus monticola* | F | 0 | 0 | 0 | 0 | 0 | X |
| *Sphyraena barracuda* | MS | 0 | X | 0 | 0 | X | X |
| *Polydactilus octenemus* | MS | X | X | X | X | 0 | 0 |

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**Table 1 (continued)**

| Species | Ecological Guild | 1966–1968 | 1987–1988 | 1989 | 1989–1990 | 1990–1991 | 2000–2001 |
|---|---|---|---|---|---|---|---|
| *Lupinoblennius nicholsi* | E | X | 0 | 0 | 0 | 0 | 0 |
| *Gobionellus oceanicus* | E | 0 | X | X | X | X | X |
| *Gobioides broussonetii* | E | X | 0 | 0 | X | X | X |
| *Lophogobius cyprinoides* | E | 0 | 0 | 0 | 0 | 0 | X |
| *Bathygobius soporator* | E | X | X | 0 | 0 | 0 | X |
| *Guavina guavina* | E | 0 | X | X | X | 0 | 0 |
| *Evorthodus lyricus* | E | X | X | 0 | 0 | 0 | 0 |
| *Lythripnus* sp. | E | 0 | 0 | X | 0 | 0 | 0 |
| *Gobiomorus dormitor* | E | X | 0 | 0 | 0 | X | X |
| *Dormitator maculatus* | E | X | 0 | 0 | X | X | X |
| *Eleotris pisonis* | E | 0 | 0 | 0 | 0 | 0 | X |
| *Erotelis smaragdus* | E | 0 | X | 0 | X | X | X |
| *Trichiurus lepturus* | MS | X | 0 | 0 | X | X | 0 |
| *Citharicthys spilopterus* | ME | X | X | X | X | X | X |
| *Achirus lineatus* | ME | X | X | X | X | X | X |
| *Trinectes maculatus* | ME | X | 0 | 0 | 0 | 0 | 0 |

To determine if the species composition of assemblages changed over time, we used PRIMER (v. 5) software to generate a non-metric multidimensional scaling (MDS) diagram. This diagram shows the relative similarity of fish assemblages in ordinate space based on pair-wise Bray-Curtis similarities of presence-absence data. Fish assemblages that are more similar appear closer together in the diagram.

We also used PRIMER to calculate average taxonomic distinctness ($\Delta^+$) and variation in taxonomic distinctness ($\Lambda^+$) for all survey periods. These 2 statistics are sample size-independent measures of diversity where the taxonomic distance between every pair of species in a given assemblage is the basis for determining relative diversity (Warwick and Clarke 1995). More specifically, average taxonomic distinctness ($\Delta^+$) is the mean taxonomic distance apart of all species pairs in an assemblage, and variation in taxonomic distinctness ($\Lambda^+$) is the variance of the taxonomic distances between each species pair about their mean (Clarke and Warwick 2001). A detailed description of the properties that make these 2 statistics sample-size independent, and therefore useful for extracting meaningful information from simple presence-absence data, is provided in Clarke and Warwick (2001). To calculate these statistics for each survey, the total list of species collected for all surveys was used. Based on classification from Nelson (1994), we identified the following taxonomic categories for each species: species, genus, family, order, superorder, subdivision, division, subclass, class, and grade. Each of these categories represents a "node" in determining taxonomic distances between species pairs.

We used this taxonomic species list in combination with the original presence-absence species data to run a TAXDTEST analysis in PRIMER. This analysis produces "funnel plots" where $\Delta^+$ and $\Lambda^+$ for each survey are plotted in comparison with the mean and 95% confidence limits of $\Delta^+$ and $\Lambda^+$ calculated for 1000 simulated matrices of presence-absence species data. Values of $\Delta^+$ and $\Lambda^+$ for observed data that fall outside of the 95% confidence limits represent significant differences in diversity from expected. For this TAXDTEST analysis, the weighting option of using taxonomic richness was chosen. For this option, the weighting of inter-category distances is calculated using the species richness information from the original presence-absence species data.

## RESULTS

A total of 107 fish species was collected during the 6 analyzed surveys (Table 1). Of these, 15 species occurred in every survey: Atlantic stingray (*Dasyatis sabina*), hardhead catfish (*Arius felis*), dark sea catfish (Cathoropus melanopus), Atlantic needlefish (*Strongylura marina*), shortfin molly (*Poecilia mexicana*), fat snook (*Centropomus parallelus*), flagfin mojarra (*Eucinostomus melanopterus*), rhombic mojarra (*Diapterus rhombeus*), Irish pompano (*Diapterus auratus*), striped mojarra (*Eugerres plumieri*), silver perch (*Bairdiella chrysoura*), whitemouth croaker (*Micropogonias furnieri*), white mullet (*Mugil curema*), bay whiff (*Citharicthys spilopterus*), and lined sole (*Achirus lineatus*; Table 1). A total of 37 species were collected in only a single survey over the 6

**TABLE 2**

**Frequency of species representing 4 ecological guilds (marine stenohaline, marine euryhaline, estuarine, and freshwater) collected from the Alvarado Lagoon Estuary during 6 surveys conducted over 33 years (1968–2001).**

| Survey Period | Marine Stenohaline | Marine Euryhaline | Estuarine | Freshwater |
|---|---|---|---|---|
| 1966–1968 | 16 | 26 | 8 | 7 |
| 1987–1988 | 16 | 26 | 7 | 6 |
| 1989 | 7 | 24 | 8 | 4 |
| 1989–1990 | 12 | 31 | 9 | 6 |
| 1990–1991 | 15 | 30 | 8 | 6 |
| 2000–2001 | 9 | 29 | 13 | 9 |

survey periods. Of these, 15 species occurred only in the 1966–1968 survey and may have since become extirpated from the estuary: speckled worm eel (*Myrophis punctatus*), scaled sardine (*Harengula jaguana*), southern buffalo (*Ictiobus meridionalis*), skilletfish (*Gobiesox strumosus*), an unidentified silverside (*Thyrinops* [*Atherinella* sp.), obscure swamp eel (*Ophisternon aenigmaticum*), Mexican snook (*Centropomus poeyi*), yellow jack (*Caranx bartholomei*), barred grunt (*Conodon nobilis*), silver seatrout (*Cynoscion nothus*), spotted seatrout (*Cynoscion nebulosus*), Jack Dempsey (*Cichlasoma octofasciatum*), blackstripe cichlid (*Vieja fenestrata*), highfin blenny (*Lupinoblennius nicholsi*), and hogchoker (*Trinectes maculatus*)(Table 1).

When these species were divided into ecological guilds, marine euryhaline species dominated the Alvarado Lagoon Estuary during all survey periods (Table 2). The mean number of marine euryhaline species collected across the 6 surveys was 27.67 species (range = 24–31),

whereas marine stenohaline, estuarine, and freshwater species averaged 12.50 (7–16), 8.83 (7–13), and 6.33 (4–9) species, respectively (Table 2). The occurrence frequency of fish species representing each ecological guild, though, did not change significantly among the survey periods. The chi-square deviation statistic calculated for the observed matrix ($x^2 = 8.53$) was not significantly larger than the mean value for 1000 simulated matrices ($x^2 = 138.64$; $P = 1.00$). The number of species representing each guild did not differ significantly from average over the 6 survey periods.

A non-metric multidimensional scaling (MDS) diagram based on Bray-Curtis similarities of fish species presence-absence data showed that the 1966 survey period was the least similar to the other survey periods (Figure 2). The assemblage representing the 2000–2001 survey period was somewhat separated from the 1987–1988, 1989–1990, and 1990–1991 surveys, but this separation was similar to the degree of separation from these surveys exhibited by the 1989 survey period (Figure 2). For the 1966–1968 survey, the range of Bray-Curtis inter-survey similarities was 40.4–58.6 ($n = 5$; Table 3). By comparison, the remaining range of inter-survey similarities was 61.5–81.7 ($n = 10$; Table 3).

Average taxonomic distinctness ($\Delta^+$) and variation in taxonomic distinctness ($\Lambda^+$) for all survey periods were within the simulated 95% confidence limits for expected values (Figures 3 and 4). Both the $\Delta^+$ and $\Lambda^+$ values for the 2000–2001 survey period, though, were less than the mean $\Delta^+$ and $\Lambda^+$ values for the entire species pool (Figures 3 and 4). The only other values that were less than average (and only slightly) were the $\Delta^+$ values for the 1987–1988, 1989, and 1990–1991 survey periods (Figure 3).
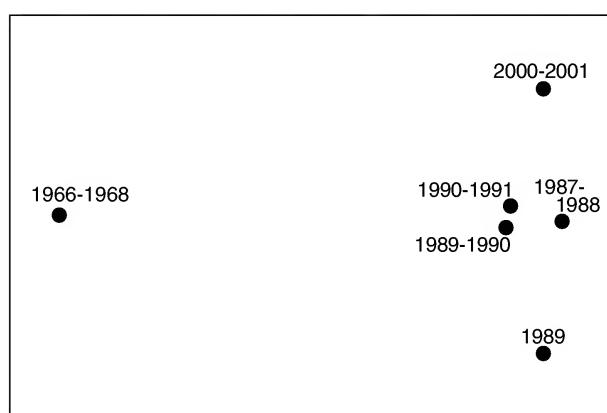


**Figure 2. Non-metric multidimensional scaling (MDS) diagram of fish species assemblage differences among the 6 fish surveys. Distances in diagram represent relative Bray-Curtis similarity values. Assemblages closer to each other are more similar. Representation of assemblage relationships is at the highest level of accuracy (stress = 0.01).**

## DISCUSSION

Fish assemblages in the Alvarado Lagoon Estuary have not changed significantly over the last 40 years. This

## TABLE 3

**Bray-Curtis inter-survey similarities for fish assemblages collected during 6 survey periods from the Alvarado Lagoon Estuary. Bray–Curtis indices typically range from 0–100 with higher values representing greater similarity between assemblage pairs.**

| Survey Period | 1966–1968 | 1987–1988 | 1989 | 1989–1990 | 1990–1991 | 2000–2001 |
|---|---|---|---|---|---|---|
| 1966–1968 | – | – | – | – | – | – |
| 1987–1988 | 50.0 | – | – | – | – | – |
| 1989 | 40.4 | 70.1 | – | – | – | – |
| 1989–1990 | 56.6 | 73.9 | 69.4 | – | – | – |
| 1990–1991 | 58.6 | 78.9 | 67.3 | 81.7 | – | – |
| 2000–2001 | 55.5 | 61.5 | 61.5 | 64.4 | 72.7 | – |

conclusion, though, needs to be considered in context of the loss of several species since 1966–1968 and an apparent more recent (since 1991) overall decline in biodiversity in this ecosystem. Unfortunately, the highly variable nature of estuarine ecosystems precludes simple diagnoses of significant changes in fish assemblages (O'Connell et al. 2004). In these ecosystems, inter-habitat movement, especially by migrating estuarine fishes, creates temporally dynamic fish faunas that are difficult to accurately assess without complete long-term data (Thompson and Fitzhugh 1985; Peterson and Ross 1991; Poff and Allan 1995). While the data and analyses presented here cannot definitively show a statistically significant change in fish-

es relative to ecological guilds or assemblages, the results suggest past and potentially future ecological changes in the fishes of this estuary.

The consistent occurrence of marine euryhaline fishes in collections over time reflects the salinity-tolerant nature of this ecological guild. Of the group of 15 species that were collected from all 6 surveys, 12 were marine euryhaline. The remaining 3 species consistently collected were *C. melanopus* (estuarine/freshwater), *P. mexicana* (freshwater), and *E. plumieri* (marine stenohaline). Marine euryhaline fishes were the most suited to withstand the variety of events that have influenced the hydrological dynamics of Alvarado Lagoon Estuary since 1966. An artificial inlet
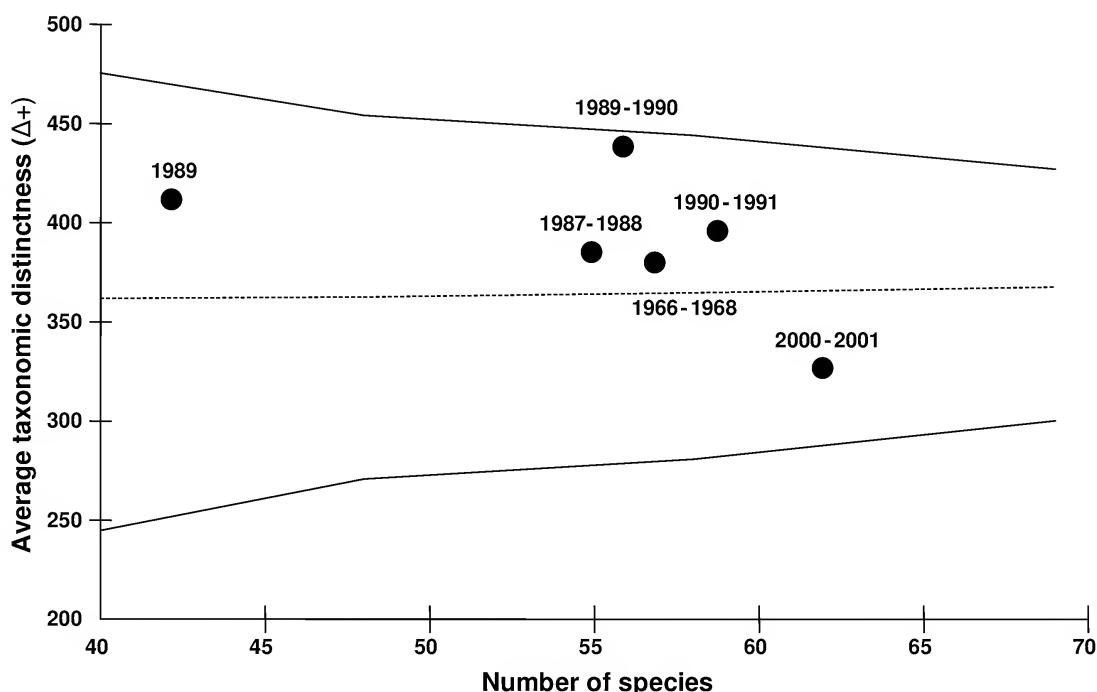


**Figure 3.** Average taxonomic distinctness ($\Delta^+$) of fish assemblages collected during the 6 surveys relative to the mean $\Delta^+$ (dotted line) and the 95% confidence intervals (solid lines) for 1000 simulated fish assemblages. Simulated fish assemblages were generated from a total species list representing all fishes collected over all surveys.

was opened in Camaronera Lagoon in 1979 with hopes of increasing the salinity in this area of the system to increase shrimp production. Rosales-Hoz et al. (1986) reported a change from 4 to 25 psu in Camaronera Lagoon after the opening of the artificial inlet. Villalobos et al. (1975) found that discharge from the rivers was the principal influence on the hydrological and salinity patterns in the 1960s. Stratification of the system occurred from the estuarine zone of the Papaloapan River to the central region of the Alvarado Lagoon. The lagoon was almost entirely oligohaline in the 1960's, with slight salinity increases provided by the tides during the dry season. In contrast, during the 1980s, the majority of Camaronera Lagoon was mesohaline (Raz-Guzmán et al. 1992). The artificial inlet at Camaronera Lagoon was dredged in 1990, but by 1996 excessive sedimentation began, greatly restricting the circulation of marine waters into the lagoon. An El Niño event in 1998 resulted in increased freshwater inflow into the lagoonal ecosystem, as was reported for other coastal lagoons (García et al. 2001, Kupschus and Tremain 2001, Mol et al. 2001). Thus, during this time period a large part of the lagoonal ecosystem had oligohaline and freshwater characteristics, with mesohaline conditions only found in the dry season near the inlets. These conditions were similar to those reported in the 1960s (Reséndez 1973; Villalobos et al. 1975). Given this variability, marine eury-

haline fishes have a considerable advantage over fishes in the other 3 ecological guilds. Neither freshwater nor marine stenohaline fishes could consistently withstand such changes in salinity. Estuarine fishes, though capable of tolerating a wide range of salinities, would be more prone than marine euryhaline species to local anthropogenic disturbances such as the opening of the artificial inlet and subsequent dredging. The entirety of their life cycles occurs in closer proximity to these impacts than marine euryhaline fishes. A similar response was seen in the Lake Pontchartrain Estuary, another degraded Gulf of Mexico ecosystem. Atlantic croaker (*Micropogonias undulatus*), an estuarine species, experienced relatively greater declines than other fishes during a period of increased local shell dredging. More transient marine species that used the estuary less frequently, though, were not as impacted (O'Connell et al. 2004).

The 15 fish species that were never collected after the 1966–1968 survey reflect the extent to which Alvarado Lagoon Estuary has changed over nearly 3 decades of multiple anthropogenic impacts. By comparison, the degraded Lake Pontchartrain Estuary lost only 3 species between 1954 and 2000 (O'Connell et al. 2004). While some of these "lost" species may be fishes that under normal conditions rarely occur in the estuary (e.g., there was a total of 37 species that were collected in only a single survey), it is
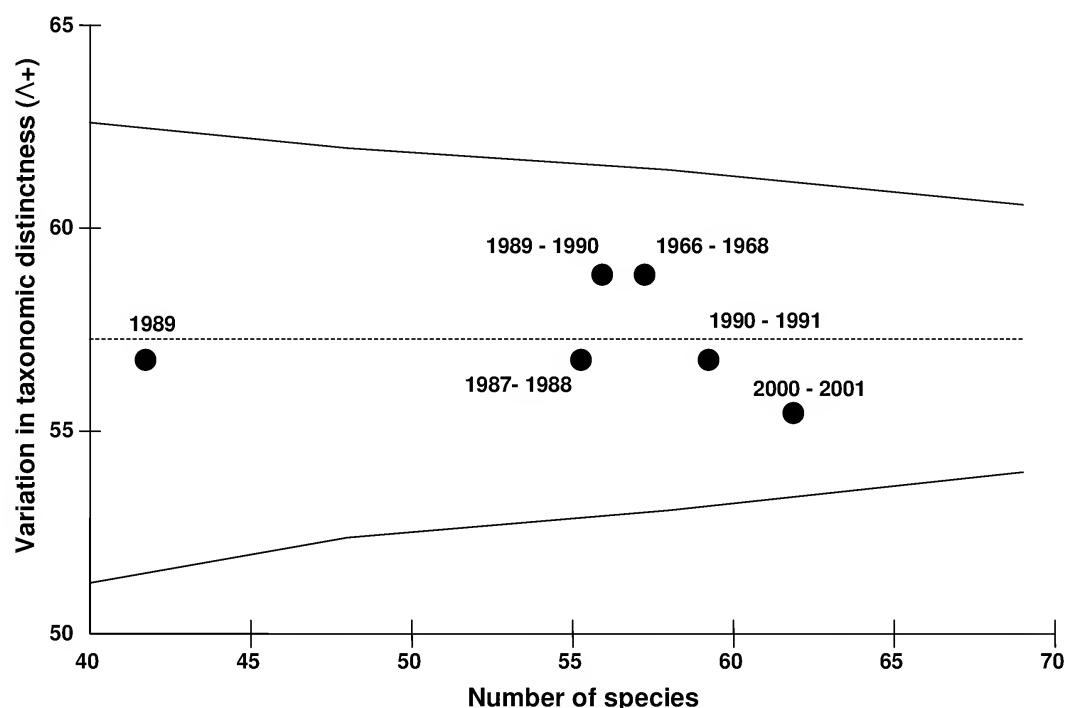


**Figure 4.** Variation in taxonomic distinctness ($\Lambda^+$) of fish assemblages collected during the 6 surveys relative to the mean $\Lambda^+$ (dotted line) and the 95% confidence intervals (solid lines) for 1000 simulated fish assemblages. Simulated fish assemblages were generated from a total species list representing all fishes collected over all surveys.

noteworthy that the survey with the most of these single occurrences was 1966–1968 (Table 1). The fact that this group of 15 single-occurrence species contains members of all 4 ecological guilds (6 marine stenohaline, 3 marine euryhaline, one estuarine, and 5 freshwater) suggests no single cause can explain the possible extirpations. For example, the loss of the freshwater cichlid species *C. octofasciatum* and *V. fenestrata* would more likely be related to habitat alteration in nearby rivers, while the absence of marine stenohaline species such as *H. jaguana* and *C. bartholomei* in later surveys may reflect responses to local salinity changes.

This loss of species over time was not enough to significantly change the occurrence frequency of species in each of the 4 guilds. The marine euryhaline guild consistently had the greatest species representation in the Estuary while the number of species for the remaining 3 guilds fluctuated at lower species numbers; the highest number of non-marine euryhaline species in any one survey period was 16 (marine stenohaline in 1966–1968 and 1987–1988) while the lowest number of marine euryhaline species in any one period was 24 in 1989. A lack of significant change over time in the numbers of species in these guilds reflects the stability and tolerance of the dominant group, the marine euryhaline species. Whether natural or anthropogenic factors are influencing fish assemblages, the ecological guilds that will respond most closely to the impacts in this estuary happen to possess fewer species. Thus, if and when degradation starts to affect fishes, we should not expect to notice a significant change by examining the system using a broad-scale approach such as comparing ecological guilds. Had we discovered significant changes at this level of analysis, we could assume a much more severe impact had affected this ecosystem.

Using the more precise approach of MDS, though, we developed a clearer understanding of how similar the surveys were to each other and how this loss of species affected the assemblages. In the MDS diagram (Figure 2), the 1966–1968 survey clearly stands apart from the other 5 surveys and this is supported by the Bray-Curtis similarity index data (Table 3). The long horizontal "leap" from the single survey on the left of the diagram to the clump of surveys on the right indicates the largest assemblage change occurred between the 1966–1968 and 1987–1988 survey periods (Clarke and Warwick 2001). For the 5 more recent surveys, it appears that the estuarine assemblages have reached a new compositional "mean" and any assemblage changes since 1987 seem centered about this mode. This "cyclicity" (Matthews 1998) in later surveys implies 2 situations: 1) recent assemblages have stabilized at a species compositional mode that is different from 1966–1968 and

2) recovery to an assemblage like that collected in the 1966–1968 survey is unlikely without massive restoration efforts. It should also be noted that within the cyclicity of the later 5 surveys, the 1989 and 2000–2001 surveys appear the furthest from the implied mode (in the diagram, 1989 is below the mode, 2000-2001 is above). While these positions might only reflect annual differences in species composition (e.g., a low rainfall year attracting more marine stenohaline fishes into the estuary), the position of the 2000–2001 assemblage may reflect the beginning of yet another compositional shift as occurred between 1966–1968 and 1987–1988. Further surveys could confirm whether the estuarine fish assemblage has stabilized (i.e., exhibits cyclicity) or is changing again (i.e., moving away from the recent compositional mode).

When compared with the other 5 surveys, the $\Delta^+$ and $\Lambda^+$ values for the 2000–2001 period indicate that assemblage diversity is decreasing, though the change is not yet significant (Figures 3 and 4). In the funnel plot diagrams this latest survey is the only period where both $\Delta^+$ and $\Lambda^+$ values are less than the calculated overall mean values (Figures 3 and 4). The implication is that the estuarine assemblage is at the beginning stages of yet another compositional change that involves the loss of species diversity. The relatively (though not significantly) depressed $\Delta^+$ value for the 2000–2001 period translates into an assemblage that is less taxonomically diverse than the other assemblages (e.g., fewer species per genus, fewer genera per family, etc.). Measuring and comparing $\Lambda^+$ (which is the variation of $\Delta^+$) allows for an even finer analysis of relative diversity. It is possible that 2 assemblages will have similar $\Delta^+$ values even when one has mostly species-rich genera while the other has many higher taxa (e.g., families, orders, etc.) represented by only one or a few species (Clarke and Warwick 2001). Therefore, when both $\Delta^+$ and $\Lambda^+$ values are relatively low (as for the 2000–2001 period) it suggests a reduction in both the normal array of higher taxa (reduced $\Delta^+$) and a loss of those higher taxa with only a few representative species in the assemblage (reduced $\Lambda^+$). As with the MDS results, the fact that this latest survey reflects a unique situation of lowered diversity relative to previous surveys should raise concerns that the fish assemblage in Alvarado Lagoon Estuary may once again be irreversibly transforming to another compositional mode.

This work represents one of relatively few published studies on fish assemblages from the southern Gulf of Mexico (Castañeda and Contreras 1994). Information on species composition from other coastal lagoons and estuaries in Mexico shows that the Alvarado Lagoon Estuary is typical, with only 3 other lagoon systems possessing high-

er fish diversity (Castañeda and Contreras 1994; Pérez-Hernández and Torres-Orozco 2000; Raz-Guzmán and Huidobro 2002). In general, the Gulf of Mexico in this region is subjected to a variety of impacts, particularly the region in south-central Veracruz. Oil and gas exploration began on the continental shelf off Alvarado in 2000 and has since moved steadily closer to the lagoon ecosystem. Population growth, changes in land use practices, an increase in the cattle industry, and unregulated fishing have negatively impacted the lagoon ecosystem, which has resulted in the disappearance of valuable habitats for fishes. Furthermore, large-scale climatic phenomena like El Niño effects and Global Warming (Blaber 2002, Whitfield and Elliot 2002) have influenced the hydrological characteristics of the system as well. The combination of these effects will no doubt continue to result in the deterioration of the integrity of the habitats of the lagoon system and the species that occupy them. Thus, this study documenting changes in assemblage composition of fishes over a 40-year period may be an important baseline data for future comparisons documenting additional anthropogenic changes in the Alvarado Lagoon Estuary.

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## Trophic Relationships of Demersal Fishes in the Shrimping Zone Off Alvarado Lagoon, Veracruz, Mexico

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# TROPHIC RELATIONSHIPS OF DEMERSAL FISHES IN THE SHRIMPING ZONE OFF ALVARADO LAGOON, VERACRUZ, MEXICO

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*ABSTRACT* The diet of demersal piscivorous fishes captured as bycatch of the commercial shrimping fleet off the
Alvarado lagoonal system, Veracruz, Mexico, was studied. Nine collections distributed throughout the nortes
(windy), wet, and dry seasons were made from November 1993 to January 1995. Sampling yielded a total of 646
fishes representing 10 families and 14 species, of which 44.9% had empty digestive tracts and were excluded from
analysis. *Trichiurus lepturus* and *Synodus foetens* were the most abundant demersal predators in the collections.
Differences in food consumption of the 7 most abundant predators were observed among the 3 seasons, with the
greatest variety of prey (20 species) taken during the nortes season and the lowest variety (9 species) during the dry
season. Five distinct trophic guilds were determined based on an index of relative importance of prey. Prey type and
location of prey within the water column helped determine guild classification. The occurrence of different trophic
guilds may allow for decreased competition for food resources on the continental shelf off Alvarado, Mexico.

*RESUMEN* Se estudio la dieta de los peces piscívoros demersales capturados como fauna acompañante del camarón
en la flota de barcos camaroneros del sistema de lagunas de Alvarado, Veracruz, México. Se obtuvieron nueve cole-
cectas que abarcaron las temporadas de nortes, lluvias y secas desde noviembre de 1993 hasta enero de 1995. Las
muestras produjeron un total de 646 peces representados por 10 familias y 14 especies; 44.9% fueron encantrados
con el tracto estomacal vacíos y no fueron analizados. *Trichiurus lepturus* y *Synodos foetens* fueron los
depredadores demersales más abundantes en nuestras muestras. Se observaron diferencias en el consumo de alimen-
to en las tres temporadas. La temporada de nortes mostró la mayor variación de presa (20 especies), y la menor
variación se observó en la temporada de secas (9 especies). Cinco distintivos gremios tróficos fueron identificados
basados en el índice de importancia relativa de la presa. El tipo de presa y la localización de las presas en la colum-
na de agua permitieron determinar la clasificación de los gremios. La existencia de diferentes gremios tróficos per-
mite una disminución en la competencia por recursos alimenticios en la plataforma continental del Alvarado,
México.

## INTRODUCTION

Shrimp trawling is one of the most important fishing industries in Mexico. In the southern Gulf of Mexico off Veracruz, a serious decline in the Mexican shrimping industry was observed from 1980 to 1991. In 1980, the shrimping industry reported a production of 5000 metric tons/year of penaeid shrimp (Grande and Díaz 1981), whereas in 1991 production using the same capture effort was only 1500 metric tons/year (SEMARNAP 1997). Currently, catches oscillate between 2000 and 3000 metric tons/year off Veracruz (Uribe-Marinez 2003). Worldwide, overfishing both by commercial and recreational fishers has reduced the abundance and biomass of apex predator species (Tegner and Dayton 1999, Jackson et al. 2001, Coleman et al. 2004) as well as non-targeted species (Burrage et al. 1993, Steele et al. 2001), leading to altered food webs in estuaries, coral reefs, and kelp forests (Jackson et al. 2001).

Data from several localities of the world show that in some types of fisheries more than 90% of the total catch (biomass) is discarded as waste bycatch (Alverson et al. 1994, Erzini et al. 2001, Kennelly and Broadhurst 2002). Studies have shown that the fish to shrimp ratio in temperate and subtropical areas of Mexico is 5:1 metric tons/yr, while the ratio in tropical areas is 10:1 metric tons/yr (Grande and Díaz 1981). Furthermore, shrimp trawling disturbs extensive areas of benthic habitat, affects the benthic macrofauna, and dramatically changes the diversity and abundance of demersal fish fauna (Alverson et al. 1994, Kaiser 1998, Rogers et al. 1999).

Little is known about the trophic structure and other ecological processes of the biotic community in the shrimping area off the Alvarado Lagoon, Veracruz, Mexico. This study was designed to examine the abundance and trophic interactions of demersal predatory fishes that are part of the bycatch in this area of high shrimp trawling effort. A common method of establishing trophic

structure is by the determination of trophic guilds (Luczkovich et al. 2002). Trophic guilds, defined as the grouping of species that share similar resources in a competitive complex (Root 1973, Blondel 2003), were determined in this study through analysis of stomach contents of trawl-caught fishes.

## METHODS

### Study Area

The study area is located immediately offshore of the Alvarado Lagoon system in the central portion of the state of Veracruz, Mexico, between 18°45'N, 95°40'W and 19°00'N, 95°42'W. Three well defined seasons characterize the region: the wet season from June through September, the nortes (windy) season from October through January, and the dry season from February through May (Contreras 1985). Highest precipitation occurs during the rainy season and oscillates between 1100–2000 mm over the year (García 1973). The Alvarado area is characterized by extensive coastal vegetation including mangroves and seagrasses and a series of lagoons and rivers that brings considerable fresh water and organic matter to the continental shelf, particularly during the rainy season.

### Sample collection and processing

We collected demersal fishes, known from the literature to be piscivores, from boats of the Alvarado shrimping fleet on 9 occasions from November 1993–January 1995, covering all 3 seasons. There were 4 collections during the nortes season, 3 collections during the wet season and 2 collections during the dry season. Boats in the fleet were equipped with a 20 m beam trawl with a 5.5 m mouth opening that was constructed with 3.85 cm mesh. Towing speed was 5–6 km/h, covering a distance of 1.8–18.5 km per sampling event. Fishing depths ranged from 30–90 m, with a mean depth of 50 m. A 30 l subsample of the bycatch (representing 25–27 kg of fish) was obtained using the methods described by Guzmán (1991) and Peláez-Rodríguez (1993) from trawls fished for 4 h between 0800–0730 local time (Central Time Zone).

Formaldehyde (10%) was injected into the oral and anal areas and then fish were immersed in the formaldehyde solution (Laevastu 1971). Fishes were labeled, bagged, and transported to the laboratory where samples were rinsed with tap water and preserved in 70% methanol within 48 to 72 hours. Species were identified with Hoese and Moore (1977), Fisher (1978), and Castro-Aguirre (1978). Fish were measured (standard length, SL, mm) and weighed to the nearest 0.1 g. Stomachs were extracted, and their contents were identified to the lowest possible taxon using hard parts such as otoliths, scales, jaw bones and cranial bones (Windell and Stephen 1978). Prey items were blotted with desiccant paper and weighed to the nearest 0.001 g; empty stomachs were noted but not included in the analysis. Stomach contents of the 7 most abundant predators captured were used for analysis. Prey items were classified as pelagic, benthic, or benthic-pelagic according to knowledge of their general occurrence within the water column (Carpenter 2002).

### Data analysis

Abundance and biomass of the predator species were compared among seasons for each subsample with analysis of variance (ANOVA) and pairwise Sidak post-hoc tests to separate mean values if a significant F-test was determined. Species richness (S) was determined seasonally based on the abundance of the demersal, predatory fishes captured. Additionally, percent contribution of each species in terms of abundance and biomass were calculated by season.

The importance of each prey species for each of the 7 most abundant fishes was evaluated by pooling data for each season and then calculating the index of relative importance (IRI; Pinkas et al. 1971), defined as IRI = %F(%N + %W), where %F = frequency of occurrence of a food item, %N = numerical percentage of a food item in the stomachs, and %W = percentage by volume of the food item in the stomachs (Pinkas et al. 1971). IRI values were standardized to %IRI for comparison (Cortés 1997).

A Bray-Curtis dissimilarity matrix was calculated based on %IRI values, and this matrix was used to construct a dendrogram using the unpaired grouping mean average (UPGMA) method (Field et al. 1982). ANOVA was calculated using SPSS (SPSS Inc, ver 11.5, Chicago, IL). Values were considered significantly different if $P < 0.05$.

## RESULTS

### Predator abundance and seasonality

Fourteen species of demersal fishes belonging to 10 families were collected during the study, yielding a total of 646 individuals with a total biomass of 54 kg (Table 1). The families Synodontidae (4 species) and Sciaenidae (2 species) contributed almost half of the total species. Of the total catch, only 362 fishes or 56.1% contained prey in their stomachs. Three species have not been previously reported for the Alvarado area; they include *Rachycentron canadum*, collected only during the nortes season, and *Synodus poeyi* and *Trachinocephalus myops*, reported for both the nortes and wet seasons (Table 1). Overall,

**TABLE 1**

**Composition of the demersal fish fauna collected from commercial shrimp nets off the Alvarado Lagoon system during the nortes, dry and wet seasons. Abbreviations are presented for the 7 most abundant species.**

| Species | Nortes Abundance (ind) | Nortes Biomass (g) | Dry Abundance (ind) | Dry Biomass (g) | Wet Abundance (ind) | Wet Biomass (g) | Total Abundance (ind) | Total Biomass (g) |
|---|---|---|---|---|---|---|---|---|
| Muraenidae | | | | | | | | |
| *Gymnothorax nigromarginatus* | 5 | 685.3 | 7 | 635.9 | 6 | 709.1 | 18 | 2030.3 |
| Ophichthidae | | | | | | | | |
| *Myrophis punctatus* | 2 | 147.0 | 3 | 192.5 | 6 | 334.7 | 11 | 674.2 |
| Synodontidae | | | | | | | | |
| *Synodus foetens* (Syfo) | 67 | 14811.7 | 32 | 3119.3 | 25 | 1761.0 | 124 | 19692.0 |
| *Synodus poeyi* | 25 | 868.4 | | | 32 | 717.2 | 57 | 1585.6 |
| *Trachinocephalus myops* | 8 | 369.4 | | | 15 | 849.8 | 23 | 1219.2 |
| *Saurida brasiliensis* (Sabr) | 15 | 75.7 | 5 | 47.9 | 66 | 291.5 | 86 | 415.1 |
| Fistulariidae | | | | | | | | |
| *Fistularia tabacaria* | 2 | 84.1 | | | 2 | 42.1 | 4 | 126.2 |
| Priacanthidae | | | | | | | | |
| *Priacanthus arenatus* | 15 | 1518.6 | 2 | 373.3 | 6 | 568.7 | 23 | 2460.6 |
| Rachycentridae | | | | | | | | |
| *Rachycentron canadum* | 2 | 1208.0 | | | | | 2 | 1208.0 |
| Sciaenidae | | | | | | | | |
| *Cynoscion arenarius* (Cyar) | 5 | 639.3 | 8 | 560.9 | 6 | 1100.6 | 19 | 2300.8 |
| *Cynoscion nothus* (Cyno) | 25 | 2316.5 | 13 | 619.5 | 22 | 1853.7 | 60 | 4789.7 |
| Sphyraenidae | | | | | | | | |
| *Sphyraena guachancho* (Spgu) | 5 | 586.0 | 10 | 445.5 | 13 | 1612.7 | 28 | 2644.2 |
| Trichiuridae | | | | | | | | |
| *Trichiurus lepturus* (Trle) | 43 | 4154.0 | 19 | 1514.9 | 87 | 6700.5 | 149 | 12369.4 |
| Scombridae | | | | | | | | |
| *Scomberomorus cavalla* (Scca) | 31 | 540.0 | 6 | 1449.2 | 5 | 315.2 | 42 | 2304.4 |
| Totals | 250 | 28004.0 | 174 | 8958.9 | 291 | 16856.8 | 646 | 53819.7 |
| Species collected | 14 | | 10 | | 13 | | 14 | |

*Trichiurus lepturus* was the most common predator species captured during the study, with a total of 149 individuals, and was the dominant species during the wet season. *Synodus foetens* and *Saurida brasiliensis* were the second and third most abundant predatory fishes captured, while *Cynoscion nothus, Scomberomous cavalla, Spyraena guachancho* and *C. arenarius* rounded out the top 7 species (Table 1).

The wet season showed the highest abundance of predatory fishes in the shrimp bycatch, but it ranked second in biomass, with 291 specimens and 17 kg. The nortes season accumulated the highest biomass of bycatch predators, 28 kg, but occupied the second place in predator fish abundance with 250 specimens. The lowest values of abundance and biomass were found during the dry season with a total of 174 specimens that yielded 9 kg (Table 1).

However, there were no significant differences among seasons for either abundance (ANOVA, $F_{2,6} = 3.46$, $P = 0.100$) or biomass (ANOVA, $F_{2,6} = 0.95$, $P = 0.438$), suggesting a relatively stable and constant bycatch of predatory fishes in the shrimp trawl fishery in the area.

Seasonally, richness of predatory bycatch fishes was greater in the nortes season followed by wet and then dry seasons (Table 1); a similar pattern was seen in total abundance as well. *Synodus foetens, T. lepturus* and *S. brasiliensis* were important contributors numerically and/or in terms of biomass to the total species complement (Tables 1 and 2). *Synodus foetens* was first and *T. lepturus* second in the nortes and dry seasons in terms of abundance and biomass. In the wet season *T. lepturus* and *S. brasiliensis* were the first and 2nd most abundant species, whereas *T. lepturus* and *C. nothus* contributed more to biomass (Table 2).

TABLE 2

**Percent contribution of abundant predatory fishes by season in terms of abundance and biomass in the shrimping zone off the Alvarado Lagoon, Veracruz, Mexico.**

| Species | Nortes | | Dry | | Wet | |
|---|---|---|---|---|---|---|
| | Abundance | Biomass | Abundance | Biomass | Abundance | Biomass |
| Synodus foetens | 26.80 | 52.89 | 30.48 | 34.82 | 8.59 | 10.45 |
| Trichiurus lepturus | 17.20 | 14.83 | 18.09 | 16.91 | 29.90 | 39.75 |
| Cynoscion nothus | 10.00 | 8.27 | 12.38 | 6.91 | 7.56 | 10.99 |
| Scomberomorus cavalla | 12.40 | 1.93 | 5.71 | 16.18 | 1.72 | 1.87 |
| Saurida brasiilensis | 6.00 | 0.27 | 4.76 | 0.53 | 22.68 | 1.73 |
| Sphyraena guachancho | 2.00 | 2.09 | 9.52 | 4.97 | 4.47 | 9.57 |
| Cynoscion arenarius | 2.00 | 2.28 | 7.62 | 6.26 | 2.06 | 6.53 |
| Synodus poeyi | 10.00 | 3.10 | | | 10.99 | 4.25 |
| Priancanthus arenatus | 6.00 | 5.42 | 1.90 | 4.17 | 2.06 | 3.37 |
| Gymnothorax nigromarginatus | 2.00 | 2.45 | 6.67 | 7.10 | 2.06 | 4.21 |
| Trachinocephalus myops | 3.20 | 1.32 | | | 5.15 | 5.04 |
| Myrophis punctatus | 0.80 | 0.52 | 2.86 | 2.15 | 2.06 | 1.99 |
| Fistularia tabacaria | 0.80 | 0.30 | | | 0.69 | 0.25 |
| Rachycentron canadum | 0.80 | 4.31 | | | | |

**Predator size and diets**

The modal SL for *S. foetens* was smaller in the nortes season than in the dry or wet seasons, whereas modal SL for *T. lepturus* was largest during the wet season (Table 3). However, the range in sizes for these 2 species overlapped for all 3 seasons. The modal SL for *S. guachancho* was larger in the nortes season compared to the dry or wet seasons, and the size range for the nortes season did not overlap with the other 2 seasons (Table 3). The remaining 4 predator size ranges and modal SL did not change much by season (Table 3). This suggests that potential ontogenetic diet shifts imbedded within seasons probably did not affect analyses of trophic spectrum.

Twenty-four prey species, including 20 fishes, three decapod crustaceans, and one cephalopod, were identified from the stomach contents of the top 7 predators. Among the fish prey, *Bregmaceros cantori* and *Microdesmus lanceolatus* have not been previously reported from the shelf off Alvarado Lagoon (Table 4). For the 7 predator species, the lowest number of prey types consumed (9) occurred during the dry season, while the highest number of prey types (20) was found during the nortes season. Prey types varied among predators and changed seasonally (Table 4).

*Synodus foetens* was the second most abundant predatory species overall and had the largest variety of prey, with a total of 17 taxa (Table 4). This species fed on the greatest diversity of prey during the nortes season, and its prey occurred throughout the water column (Figure 1). Fifty-one percent IRI of the prey was benthic and included

TABLE 3

**Summary statistics on fish standard length (range and mode, cm) by season for the 7 predators used in the diet analysis.**

| Species | Nortes | | Dry | | Wet | |
|---|---|---|---|---|---|---|
| | Range | Mode | Range | Mode | Range | Mode |
| Sphyraena guachancho | 29.3–34.2 | 30.0 | 16.5–19.1 | 18.0 | 18.0–20.8 | 19.0 |
| Synodus foetens | 12.3–21.7 | 18.0 | 20.6–35.7 | 29.0 | 17.6–43.5 | 32.0 |
| Trichiurus lepturus | 32.0–58.6 | 46.0 | 39.4–51.8 | 47.0 | 45.6–89.7 | 64.0 |
| Cynoscion arenarius | 18.5–23.4 | 20.0 | 17.5–19.40 | 18.0 | 18.5–24.6 | 21.0 |
| Cynoscion nothus | 15.6–19.0 | 17.0 | 14.2–18.0 | 16.0 | 16.2–21.6 | 19.0 |
| Saurida brasiliensis | 7.4–8.9 | 8.0 | 5.2–7.6 | 6.0 | 8.4–11.0 | 9.0 |
| Scomberomorus cavalla | 24.5–27.3 | 25.0 | 22.6–28.4 | 24.0 | 21.5–27.6 | 25.0 |

## TABLE 4

### Seasonal food composition and %IRI for 7 demersal fishes off Alvarado, Veracruz.

| Species | Nortes Prey type | %IRI | Dry Prey type | %IRI | Wet Prey type | %IRI |
|---|---|---|---|---|---|---|
| S. guachancho | Anchoa hepsetus | 68.44 | Bregmaceros cantori | 66.15 | Anchoa hepsetus | 22.02 |
| | Cynoscion nothus | 1.82 | Saurida brasiliensis | 8.56 | Saurida brasiliensis | 57.6 |
| | Bregmaceros cantori | 8.94 | Loligo pealei | 25.29 | Loligo pealei | 20.38 |
| | Saurida brasiliensis | 15.34 | | | | |
| | Loligo pealei | 5.46 | | | | |
| S. foetens | Anchoa hepsetus | 21.17 | Anchoa hepsetus | 51.26 | Saurida brasiliensis | 19.36 |
| | Saurida brasiliensis | 2.03 | Upeneus parvus | 48.74 | Upeneus parvus | 8.29 |
| | Upeneus parvus | 12.92 | | | Loligo pealei | 19.33 |
| | Loligo pealei | 7.44 | | | Bregmaceros cantori | 17.70 |
| | Harengula clupeola | 14.55 | | | Pristipomoides aquilonaris | 1.76 |
| | Trachurus lathami | 2.64 | | | Diplectrum bivittatum | 23.17 |
| | Micropogonias furnieri | 2.99 | | | Syacium gunteri | 3.57 |
| | Pristipomoides aquilonaris | 9.16 | | | Trichiurus lepturus | 6.24 |
| | Diplectrum bivittatum | 8.57 | | | Engyophrys senta | 0.56 |
| | Symphurus plagiusa | 1.46 | | | | |
| | Haemulon aurolineatum | 3.14 | | | | |
| | Serranus atrobranchus | 13.01 | | | | |
| | Eucinostomus gula | 0.90 | | | | |
| T. lepturus | Anchoa hepsetus | 49.48 | Upeneus parvus | 33.28 | Anchoa hepsetus | 36.43 |
| | Upeneus parvus | 15.11 | Harengula clupeola | 24.01 | Upeneus parvus | 1.19 |
| | Pristipomoides aquilonaris | 13.72 | Loligo pealei | 42.71 | Pristipomoides aquilonaris | 1.49 |
| | Harengula jaguana | 12.2 | | | Diplectrum bivittatum | 0.55 |
| | Harengula clupeola | 5.85 | | | Synodus foetens | 0.23 |
| | Loligo pealei | 2.43 | | | Bregmaceros cantori | 2.54 |
| | Farfantepenaeus sp. | 1.21 | | | Saurida brasiliensis | 8.81 |
| | | | | | Cynoscion nothus | 0.38 |
| | | | | | Myrophis punctatus | 7.19 |
| | | | | | Loligo pealei | 8.27 |
| | | | | | Farfantepenaeus sp. | 32.89 |
| C. arenarius | Saurida brasiliensis | 27.05 | Upeneus parvus | 76.21 | Saurida brasiliensis | 33.96 |
| | Upeneus parvus | 35.47 | Diplectrum bivittatum | 23.79 | Upeneus parvus | 24.11 |
| | Pristipomoides aquilonaris | 15.21 | | | Loligo pealei | 41.93 |
| | Loligo pealei | 0.65 | | | | |
| | Farfantepenaeus sp. | 21.62 | | | | |
| C. nothus | Pristipomoides aquilonaris | 37.91 | Bregmaceros cantori | 95.49 | Bregmaceros cantori | 51.59 |
| | Bregmaceros cantori | 31.92 | Farfantepenaeus sp. | 4.51 | Saurida brasiliensis | 20.41 |
| | Saurida brasiliensis | 13.15 | | | Trichiurus lepturus | 6.05 |
| | Trichiurus lepturus | 4.16 | | | Farfantepenaeus sp. | 21.94 |
| | Microdesmus lanceolatus | 0.15 | | | | |
| | Loligo pealei | 12.70 | | | | |
| S. brasiliensis | Bregmaceros cantori | 78.42 | Bregmaceros cantori | 67.62 | Bregmaceros cantori | 75.16 |
| | Loligo pealei | 21.58 | Loligo pealei | 32.38 | Loligo pealei | 24.84 |
| S. cavalla | Anchoa hepsetus | 94.08 | Anchoa hepsetus | 69.67 | Anchoa hepsetus | 25.52 |
| | Bregmaceros cantori | 5.92 | Upeneus parvus | 30.33 | Diplectrum bivittatum | 53.52 |
| | | | | | Loligo pealei | 20.95 |
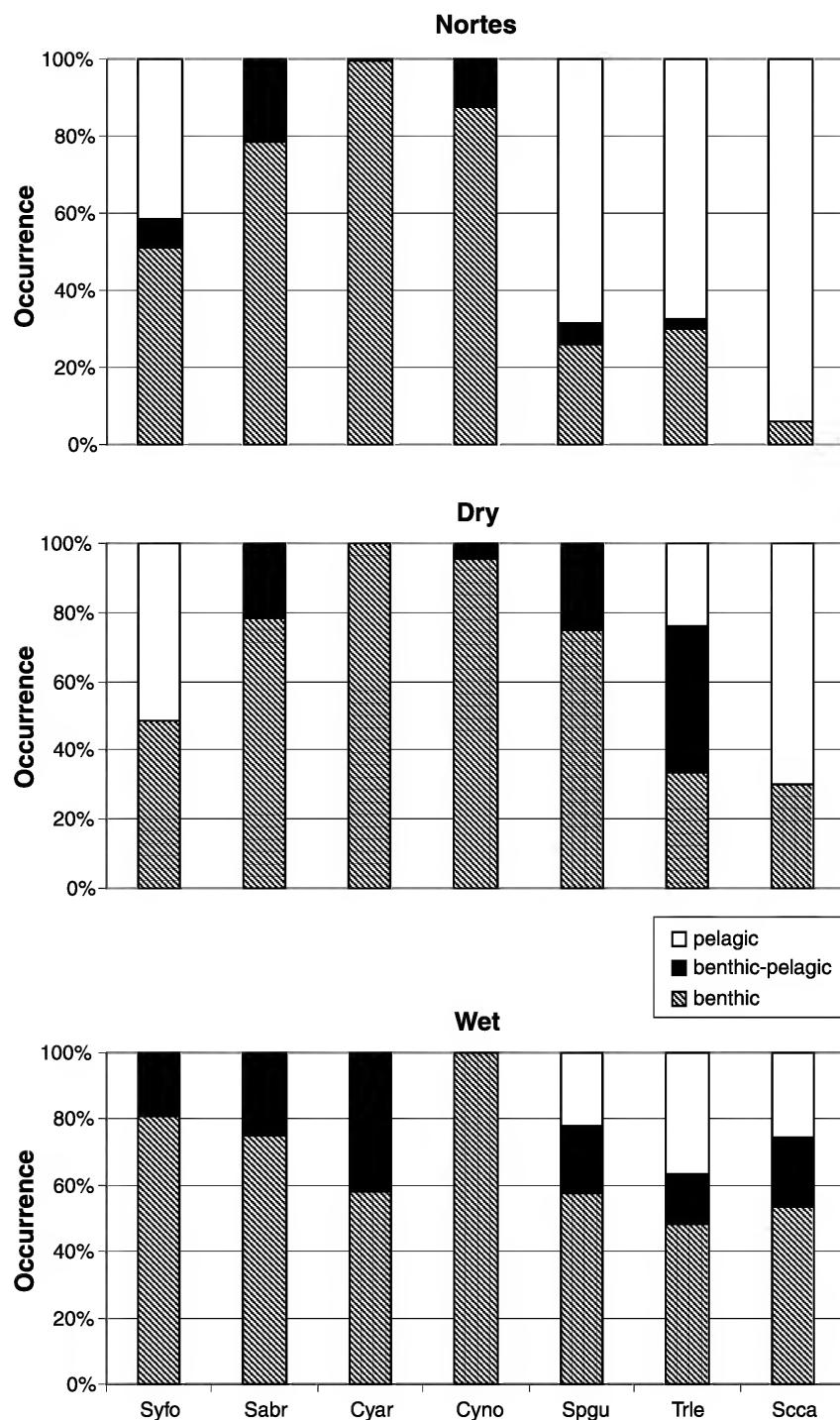
Figure 1. Percentage of prey occurring in the pelagic, benthic-pelagic, and benthic zones of the water column for 7 demersal fishes off the Alvarado Lagoon system, Veracruz, Mexico, during the nortes, dry, and wet seasons. *Synodus foetens* (Syfo), *Saurida brasiliensis* (Sabr), *Cynoscion arenarius* (Cyar), *Cynoscion nothus* (Cyno), *Sphyraena guachancbo* (Spgu), *Trichiurus lepturus* (Trle), *Scomberomorus cavalla* (Scca). Sample size for each species in the figure by season is found in Table 1.

*Upeneus parvus, Diplectrum bivittatum, Pristipomoides aquilonaris, Eucinostomus gula, Micropogonias furnieri, Symphurus plagiusa*, and *S. brasiliensis*. Species from the pelagic zone contributed 41.5 %IRI of stomach contents and included *Anchoa hepsetus, Harengula clupeola*, and *Trachurus lathami*. A smaller percentage of the diet, 10.5 %IRI, was composed of *Loligo pealei* and *Haemulon aurollineatum* from the benthic-pelagic zone. During the dry season, *S. foetens* fed about equally on *A. hepsetus* from the pelagic zone and on *U. parvus*, a bottom dweller. During the wet season, the diet of *S. foetens* was dominated by benthic prey (80.6 %IRI) which included *S. brasiliensis, D. bivittatum, T. lepturus. U. parvus, P. aquilonaris, Engyophrys senta, Syacium gunteri*, and *B. cantori*. The benthic-pelagic zone contributed 19.3 %IRI to the diet; the only prey was *L. pealei* (Table 4).

*Saurida brasiliensis* was the smallest piscivorous predator in the study and showed no differences in seasonal prey consumption (Table 4, Figure 1). This species also had the least diverse diet, with the benthic *B. cantori* accounting for 67–78 %IRI of the diet each season. A benthic-pelagic species, *L. pealei,* made up the rest of the diet (Table 4).

Neither *Cynoscion arenarius* nor *C. nothus* consumed any pelagic prey during the course of this study (Figure 1). Both species had the greatest diversity of prey items during the nortes season. During both the nortes and dry seasons, > 80 %IRI of the diet was composed of benthic species such as *U. parvus, Farfantepenaeus* sp., *P. aquilonaris, B. cantori*, and *S. brasiliensis*, whereas the remaining diet was composed of the benthic-pelagic *L. pealei* (Table 4). The diet of *C. arenarius* was dominated by benthic species during the wet season, as was the diet of the congener *C. nothus* (Figure 1). While the 2 *Cynoscion* species fed within the same areas of the water column, there were differences in the prey they captured. For instance, *B. cantori* was an important component of the diet of *C. nothus* throughout the year, yet this prey was never eaten by *C. arenarius* (Table 4). Similarily, *U. parvus* dominated the diet of *C. arenarius* but was never taken by *C. nothus* (Table 4).

*Sphyraena guachancho* consumed only 5 prey items, yet there was marked seasonal variation in the dominant prey items (Table 4). For instance, the pelagic *A. hepsetus* dominated the diet in the nortes season, while no pelagic species were consumed during the dry season when the benthic *B. cantori* dominated the diet (Table 4, Figure 1). During the wet season, benthic prey such as *S. brasiliensis* was dominant in the diet.

*Trichiurus lepturus* was the most abundant predator species captured during the study and the only species to feed throughout the water column year round (Figure 1).

During the wet season, benthic (48 %IRI) and pelagic (36.4 %IRI) species constituted the majority of the diet (13 species) of *T. lepturus*; prominent taxa included *Farfantepenaeus* sp., *S. brasiliensis,* and *A. hepsetus* (Table 4). In contrast, the pelagic species *A. hepsetus, Harengula jaguana,* and *H. clupeola* dominated the diet during the nortes season (67.5 %IRI). During the dry season, *T. lepturus* fed on 3 prey species, one from each section of the water column. The benthic-pelagic *L. pealei* (42.7 %RI) dominated the diet (Table 4, Figure 1).

*Scomberomorus cavalla* consumed only 5 prey types during the course of the study. The pelagic *A. hepsetus* dominated the diet during both the nortes (94.1 %IRI) and dry (69.7 %IRI) seasons and also accounted for 25 %IRI of the diet during the rainy season (Table 4). While benthic prey were taken throughout the year and dominated the diet in the rainy season (Figure 1), *S. cavalla* fed on different benthic species during each season (Table 4).

## Species/season dietary patterns

Five distinct trophic guilds were delimited (Figure 2). Fishes in feeding guild A consumed mainly pelagic prey like *A. hepsetus, H. jaguana*, and *H. clupeola,* whereas fish in guild B consumed not only pelagic species but transitioned to feeding on benthic-pelagic species like *L. pealei* (Figure 2). Fishes in guild C were characterized by feeding on a mixture of benthic-pelagic and benthic prey like *Farfantepenaeus* sp., *S. brasiliensis, U. parvus, Myrophis punctatus*, and *L. pealei* (Figure 2). Fishes in feeding guilds D and E tended to focus on more benthic prey like *S. brasiliensis, Farfantepenaeus* sp., *U. parvus*, and *B. cantori.*

In general, the species/season trophic patterns identified by guild analysis did not follow clear patterns, most likely due to body size-mouth gape differences and to seasonal prey availability. For example, *C. nothus, S. brasiliensis* and *C. arenarius* exhibited no seasonal differences in trophic guild, and *C. nothus* and *C. arenarius* were assigned to different guilds (Figure 2). This suggests minimal differences in prey across seasons for these species. In contrast, members of guilds A and B were comprised of different species and seasons with no clear patterns (Figure 2). Some species/season diets clustered together, and others did not. It was clear, however, that some species shifted from pelagic to benthic prey with season. For example, *S. guachancho* fed on pelagic species during the nortes and wet seasons but shifted to benthic prey during the dry season. However, the modal SL and size ranges for *S. guachancho* were virtually identical during the dry and wet seasons (Table 3), suggesting the seasonal shift in prey is not related to ontogenic feeding dif-
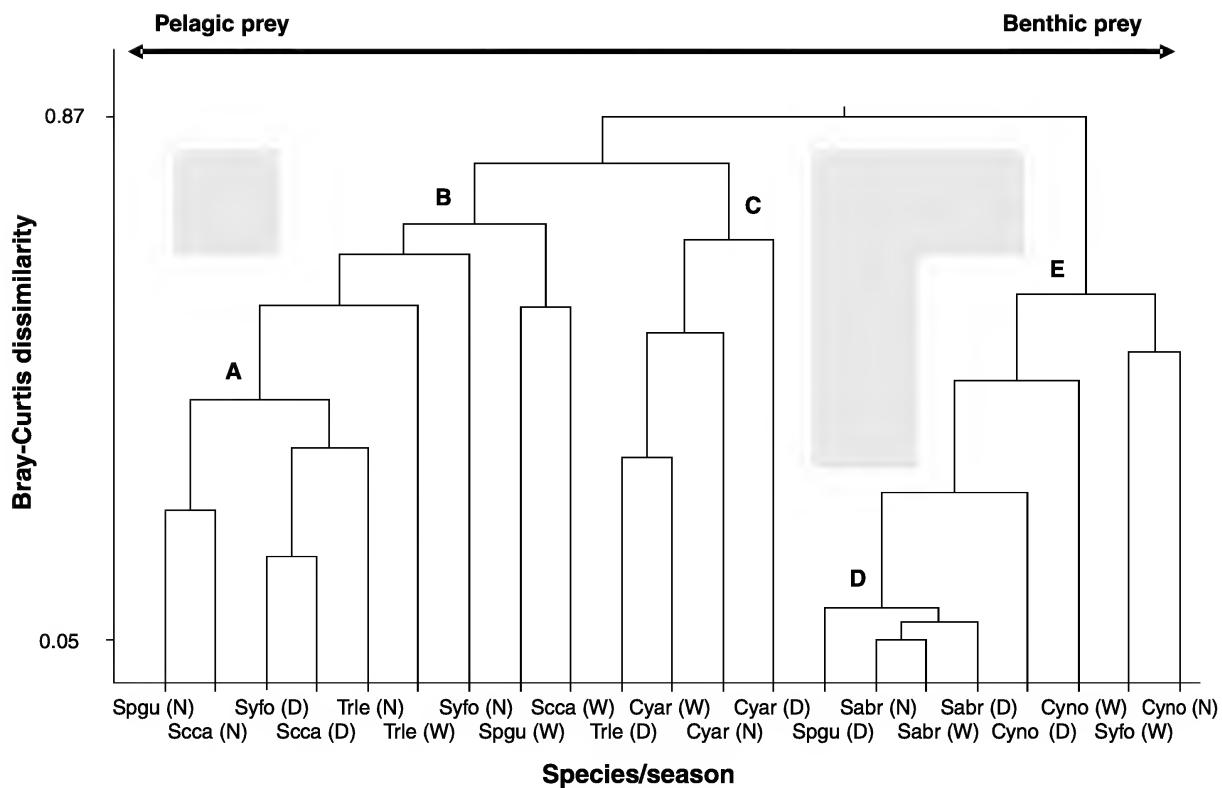
Figure 2. UPGMA cluster analysis of %IRI based on Bray Curtis dissimilarity index for 7 demersal fishes off Alvarado Lagoon Veracruz, Mexico. *Sphyraena guachancho* (Spgu), *Synodus foetens* (Syfo), *Trichiurus lepturus* (Trle), *Scomberomorus cavalla* (Scca), *Cynoscion arenarius* (Cyar), *Cynoscion nothus* (Cyno), *Saurida brasiliensis* (Sabr). N = nortes season, W = wet season, and D = dry season. Letters indicate trophic guilds identified from the cluster analysis.

ferences. Similarily, *S. foetens* and *T. lepturus* were found in 3 different guilds based on season, suggesting differences in size may not be as important as other factors determining prey selection. *Synodus foetens* fed on pelagic, benthic-pelagic, and benthic prey during all seasons, and *T. lepturus* fed on pelagic and benthic-pelagic prey. These 2 species were the most abundant species examined during this study and contributed the highest portion of biomass.

## DISCUSSION

Stomach content analysis is used widely to determine food composition, feeding strategies, trophic position, energy flow of predator and prey (Hyslop 1980), trophic structure (Luczkovich et al. 2002), and trophic partitioning (Ross 1986). Our analysis indicates the examination of stomach contents of top carnivores is an excellent way to evaluate the relationship between predators and food source in the shrimp grounds of Veracruz, Mexico.

The diets reported here for the 7 most abundant predators are generally similar to previous reports (Naughton and Saloman 1981, Mericas 1981, Divita et al. 1983,

Sheridan et al. 1984, Cruz-Escalona et al. 2005), with some notable exceptions. While fish (in particular *Anchoa*) were important in the diet of *T. lepturus* in both this study and in the northern Gulf of Mexico (GOM) (Mericas 1981, Sheridan et al. 1984), the seasonal dominance of squid in the diet (42.7 %IRI during the dry season) has not been previously reported. The diets of both *C. arenarius* and *C. nothus* captured off Veracruz differed from previous reports for the species (Sheridan et al. 1984, Sutter and McIlwain 1987) in that no pelagic prey were noted in the present study, while *Anchoa* was a major component of the diet of both species in the northern GOM (Sheridan et al. 1984). Furthermore, *Bregmoceros*, common in the diets of other predators captured in the present study, was not found in either *Cynoscion* species, although this prey species was previously reported as an important component of the diet (Sheridan et. al 1984). While the diet of *S. brasiliensis* was dominated by fish as expected, squid was a more important component of the diet of (21.5–31.3 %IRI) than the 9% frequency of occurrence previously reported by Divita et al. (1983). The predominantly piscivorous diet of *S. foetens* agrees with previous reports from the northern GOM (Divita et al. 1983) and the Veracruz,

Mexico, area (Cruz-Escalona et al. 2005), although the complete absence of penaeid shrimp in the diet is in contrast to reports from the northern GOM (Divita et al. 1983).

Our results showed patterns of resource partitioning and indicated that the 7 most abundant species examined in our study had proportioned diets based on where in the water column their prey was found. This tendency towards resource partitioning coincides with findings by Abarca-Arenas et al. (2004), who found similar evidence of resource partitioning in the Alvarado area based on the entire fish community. Macpherson (1981) and Livingston (1982) stated that in a trophic system, resource partitioning always will be observed; the pattern can be observed at the temporal level or in some cases at the diel level, even when competition among species exists

Five trophic guilds were clearly identified in our study based on the level of the water column in which the prey was obtained. Two guilds fed mainly on pelagic prey, 2 fed more on benthic prey, and one fed more on benthic-pelagic prey. Noteworthy is the large number of prey items consumed by the latter guild, demonstrating capacity to feed throughout the water column and to maintain generalist prey consumption habits.

Formation of the guilds did not appear related to body size, but rather to prey availability. Based on stomach contents, prey selection varied among the 3 seasons. Sedberry (1983) studied a community of demersal fishes on the continental shelf of the middle Atlantic Bight and also documented seasonal prey-shifting that appeared to be independent of predator size. The dry season showed the fewest taxa of prey taken, and the nortes season showed the most. With predator abundance remaining constant year-round and prey sources varying, guild structure was most likely affected. Although measurements were not made of abundance and diversity of prey beyond those obtained via stomach contents, our results suggest that prey in the nortes and wet seasons are more diverse than prey in the dry season, thus affecting the trophic guilds (sensu Darnell 1961).

Seasonal nutrient flux may influence prey availability in the study area and thus the structure of trophic guilds. Nutrients in the Alvarado Lagoon system are largely dependent upon influx from the Papaloapan River. The river deposits the largest amount of nutrients into the system during the wet and nortes seasons (Moran-Silva et al. 2005), resulting in higher productivity levels (Abarca-Arenas et al. 2004) and a general increase in the amount of exploitable resources in the system (Contreras 1985, Soberón and Yañez-Arancibia 1985). Thus, it was not a surprise that our study found the largest variety of prey and the highest abundance of predators during these 2 seasons.

Anthropogenic factors can affect the guild structure as well. Shrimp trawling is an important commercial activity off Alvarado (Grande and Díaz 1981). The effects of bycatch removal on the local demersal fish community have not been measured; however, evidence suggests that large-scale fishing affects the structure of fish communities by reducing the abundance of prey and predators and by reducing the size of predators (Pope and Knights 1982, Rice and Gislason 1996, Jennings et al. 1998, Rogers et al. 1999). In the Alvarado area, information is lacking regarding fishing activities and the life history and ecology of piscivorous fishes and their prey; thus, it is difficult to estimate the effect of the shrimp fishery and its bycatch on the trophic dynamics of the area. However, intense fishing activity in tropical waters can cause reduction in species richness and dominance of the smaller targeted and non-targeted fishes in the assemblage (Rogers et al. 1999). Our data suggest that a similar reduction in larger species may have occurred near Alvarado. For instance, large, potentially commercially important species such as *R. canadum*, *F. tabacaria*, *S. guachancho*, and *S. cavalla* composed only 11% of the total bycatch. Dominance of *S. foetens* and *T. lepturus*, 2 non-target species with the greatest variety of prey, suggests trophic adaptability and generalization may be important in this heavily fished system.

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Selected Life-History Observations on the Cayman Gambusia, *Gambusia xanthosoma* Greenfield, 1983 (Poeciliidae)

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## SHORT COMMUNICATION

# SELECTED LIFE-HISTORY OBSERVATIONS ON THE CAYMAN GAMBU-SIA, *GAMBUSIA XANTHOSOMA* GREENFIELD, 1983 (POECILIIDAE)

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### INTRODUCTION

The Cayman gambusia (*Gambusia xanthosoma* Greenfield, 1983) is an uncommon species within the *G. punctata* species group, endemic to North Sound, Grand Cayman Island, BWI. Since the original description (Greenfield 1983) only phylogenetic information has been published and little is known of its habitat, feeding ecology, or reproductive life history (Wildrick and Greenfield 1985, Rauchenberger 1988). Originally described from a brackish-water (30 psu) mosquito control ditch, the species also occurred throughout marine mangrove habitat and inland saline ponds adjacent to North Sound, Grand Cayman Island (see Figure 1). Here we present information on the habitat, diet, reproduction, life history, and parasites of the Cayman gambusia.

### MATERIALS AND METHODS

We made collections and recorded environmental data of Cayman gambusia during 4 separate collecting trips:

August 1996, January 1997, June 1997, and August 1999 (Figure 1). We recorded water temperature (°C), salinity (psu), pH, sediment type (i.e., detritus, ironshore rock, mud, sand, silt) and submerged structure (i.e., mangrove roots) at the time of sampling. Samples of Cayman gambusia were taken using either a 4.7 m long, 3 mm stretch-mesh seine; a 47 x 25 cm wide, 1 mm mesh kicknet; or a 2.5 m diameter, 1 cm mesh cast net. Presence/absence observations were made in areas inaccessible to sampling gear. Specimens were fixed in 10% Formalin, labeled, and returned to the laboratory where they were transferred to 70% ethanol.

We examined diet, reproductive characteristics, and parasites of preserved specimens from the Little Salt Creek collection made on 25 January 1997, the only collection with gravid specimens. All males were measured to the nearest 0.01 mm standard length (SL) and examined for maturity based on anal-fin morphology (Turner 1941). Males were considered mature if the formation of the gonopodium was complete.
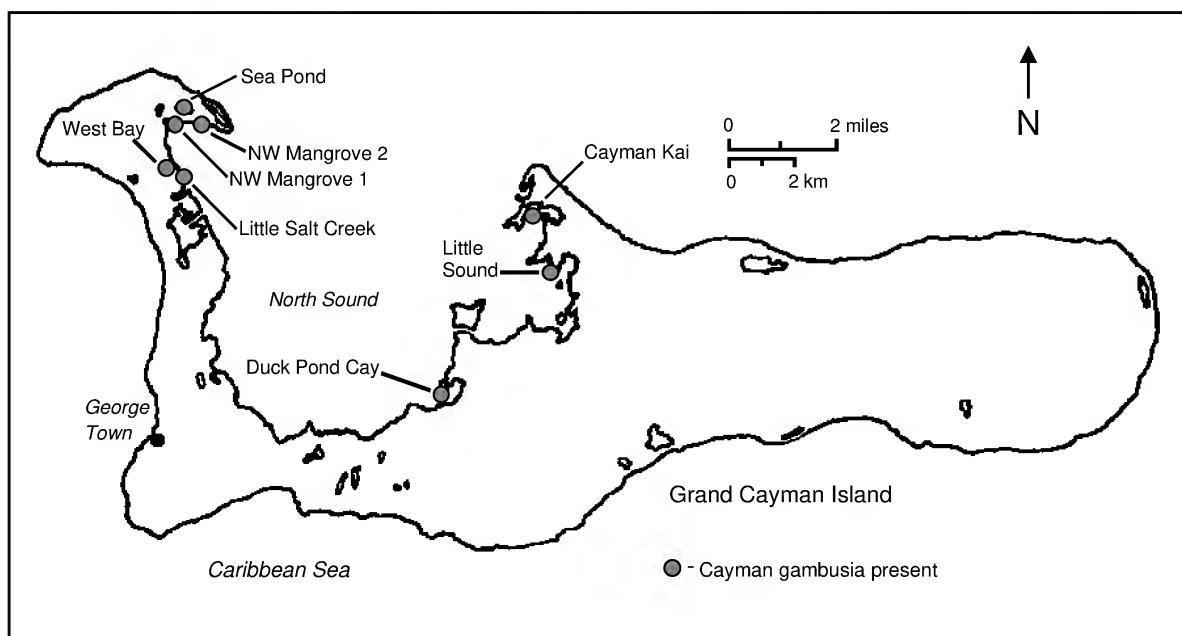


Figure 1. Locations of Cayman gambusia presence at Grand Cayman Island, BWI.

## TABLE 1

**Environmental variables recorded at the time of sampling for Cayman gambusia from each location between August 1996 and August 1999. n/a = not available.**

| Location | Date | Salinity (psu) | Temperature (°C) | D.O. (mg/l) | pH | Depth (m) |
|---|---|---|---|---|---|---|
| Cayman Kai | 8-14-99 | 39.2 | 32.6 | 3.50 | n/a | 1.2 |
| Duck Pond Cay | 8-10-99 | 37.5 | 33.9 | 5.4 | n/a | 0.3 |
| Little Salt Creek | 1-25-97 | 36.7 | 25.6 | 3.93 | n/a | 0.5 |
| Little Sound | 8-13-99 | n/a | n/a | n/a | n/a | 1.0 |
| NW Mangrove-1 | 8-12-99 | 38.2 | 32.1 | 4.5 | n/a | 0.8 |
| NW Mangrove-2 | 8-12-99 | 37.1 | 31.6 | 2.36 | n/a | 0.8 |
| Sea Pond | 8-21-96 | 34.1 | 28.3 | 4.60 | 8.0 | 1.0 |
| Sea Pond | 1-15-97 | 25.9 | 29.4 | 8.42 | 9.1 | 1.0 |
| West Bay | 6-13-97 | 30.7 | 29.6 | 1.22 | 8.3 | 0.5 |

Developmental stages were determined according to Haynes (1995), wherein ova and embryos were classed into 11 distinct morphological stages. Maturity in females was determined by the presence of a blastodisc (stage 4) or later stage embryo. Embryos of each stage and somatic tissues were dried separately overnight at 60 °C and weighed to the nearest 0.01 mg. Total embryo dry weight was determined by summing the dry weights of all embryos from stages 4 through 11. For each specimen, the mean embryo dry weight was determined by dividing the total embryo dry weight by the number of embryos in the brood. Reproductive allotment (RA), an index of the resources invested by an individual female into the production of a single brood, was estimated according to Reznick and Endler (1982). Digestive tract contents of mature females were identified to the order or class taxonomic level.

Pearson correlations were performed among 4 female reproductive traits and somatic dry weight using SPSS software (SPSS 11.0). Somatic dry weight and all reproductive traits except mean stage were $\log_{10}$ transformed prior to analysis. Significance levels were adjusted for multiple testing using the Sequential Bonferroni correction (Peres-Neto 1999). A strong negative relationship between embryo weight and mean stage of development would indicate lecithotrophy (i.e., mother does not supplement pre-fertilization yolk nutrients during embryo development), whereas a slope of zero or a shallow slope would indicate some matrotrophy (i.e., mother supplements pre-fertilization yolk nutrients during embryo development).

## RESULTS AND DISCUSSION

Based on many collections made over the course of 4 extensive surveys of ponds throughout the Cayman islands, Cayman gambusia appears to be confined to the North Sound on the western end of Grand Cayman Island, near mangroves. Specimen collection was difficult due to dense mangrove prop roots and at most locations only presence/absence could be noted. Habitat was either near or within shallow (0.5–1.0 m) fringing mangroves with muddy detritus substratum along the edges of North Sound and in associated mosquito control ditches and connected pond systems. Other fishes co-occurred including 2 poeciliid species, Cayman limia (*Limia caymanensis*) and Caribbean gambusia (*G. puncticulata puncticulata*), and various non-poeciliid species such as tarpon (*Megalops atlanticus*), hardhead silverside (*Atherinomorus stipes*), crested goby (*Lophogobius cyprinoides*), gray snapper (*Lutjanus griseus*), and sheepshead minnow (*Cyprinodon variegatus*). Based on the presence of many piscivorous wading birds (e.g., egrets, herons), avian predation was likely, although differences in the intensity of predation among locations and seasons were not known.

Physico-chemical conditions varied among the 8 collection sites and between seasons (Table 1), probably due to habitats having variable direct or indirect connections to the relatively high salinity of North Sound. The salinity ranged from 25.9 to 39.2 psu, and water temperature ranged from 25.6 to 33.9 °C.

Of the 36 males collected from Little Salt Creek on 25 January 1997, 28 were mature; SL of mature individuals ranged from 18.00 to 31.48 mm ($\bar{x}$ = 25.37 mm). Of the 58 females collected and dissected, 27 were gravid; SL of mature individuals ranged from 21.90 to 38.60 mm ($\bar{x}$ = 28.58 mm). Variation existed among gravid females in mean embryo dry weight (range 1.80–3.92 mg; $\bar{x}$ = 2.70), number of embryos per brood (range 1–11; $\bar{x}$ = 3.89) and RA (range 1.74–13.54; $\bar{x}$ = 8.27) (Table 2).

## TABLE 2

**Variation in female reproductive traits based on 27 gravid Cayman gambusia specimens from Little Salt Creek on 25 January 1997. $s_{\bar{x}}$ = standard error.**

|  | Minimum | Maximum | Mean ± $s_{\bar{x}}$ |
|---|---|---|---|
| Standard length (mm) | 21.90 | 38.60 | 28.58 ± 0.77 |
| Somatic dry weight (mg) | 46.09 | 322.26 | 118.48 ± 11.60 |
| Mean stage | 4 | 11 | 6.74 ± 0.36 |
| Mean embryo dry weight (mg) | 1.80 | 3.92 | 2.70 ± 0.11 |
| Number of embryos | 1 | 11 | 3.89 ± 0.47 |
| Total embryo dry weight (mg) | 2.73 | 25.98 | 10.20 ± 1.21 |
| Reproductive allotment | 1.74 | 13.54 | 8.27 ± 0.50 |

None of the gravid females exhibited superfetation, (i.e., the presence of multiple, non-successive developmental embryo stages, Turner 1940). Somatic dry weight was correlated with all 4 reproductive traits: larger females tended to have more and larger embryos at more advanced stages of development (Table 3). However, as a result of doing multiple statistical tests, only total embryo weight remained significantly related to somatic weight (adjusted $P < 0.05$) after Sequential Bonferroni corrections were applied. Total embryo weight was strongly correlated with the number of embryos present, even after correction.

The lack of any detectable correlation between mean embryo weight and mean stage suggested the presence of matrotrophic provisioning (Table 3; $r = 0.1$, $P > 0.25$). However, because of the limited data available, the possibility of spatio-temporal variation in facultative matrotrophy and in the amount of female provisioning still exists (Trexler 1985). Such facultative matrotrophy was observed in Caribbean gambusia (Abney and Rakocinski 2004) and Cayman limia (M.A. Abney, unpublished data).

### Diet Observations

In addition to pollen grains and seed pods, 9 taxonomic classes representing 21 orders of prey groups were identified from 27 females (Table 4). The Diptera was repre-

sented by the subgroups Brachycera, including some Cyclorrhapha, and multiple species of Nematocera. The Hemiptera was represented by multiple taxa, including the family Naucoridae. Hymenopterans included wasps (Apocrita) and ants (Formicidae). Individual species identified were the ischyrocerid amphipod *Erichthonius* cf. *brasiliensis*, the oniscid ispopod *Littorophiloscia* cf. *culebrae* (Philosciidae) and the tanaid *Hargeria rapax* (Leptocheliidae). Benthic, pelagic, and terrestrial prey were often found within the same specimen suggesting an opportunistic and generalist feeding behavior, typical of gambusiines (Meffe and Snelson 1989).

### Parasite Observations

Several helminth parasites were noted from 40 adult female Cayman gambusia (38 preserved, 2 fresh caught) at the Salt Creek site. The widely occurring and relatively non-host specific ectoparasitic monogenean, *Neobenedenia melleni* (MacCallum), was observed attached to the head of a Cayman gambusia collected 27 January 1997 (Bullard et al. 2000).

The body cavity of one Cayman gambusia contained the third stage larva of *Contracecum* sp., a nematode whose adult stage occurs in the digestive tract of piscivorous birds seen during collections. At least 2 different

## TABLE 3

**Correlations among female reproductive traits based on 27 gravid Cayman gambusia specimens from Little Salt Creek on 25 January 1997. All variables except Mean stage are correlated $\log_{10}$ scale. Bold values indicate significant before sequential Bonferroni correction; asterisks indicate significant after correction for multiple tests.**

|  | Mean stage | Mean embryo dry weight | Number of embryos | Total embryo dry weight |
|---|---|---|---|---|
| Somatic dry weight | 0.284 | **0.405** | 0.504 | **0.628*** |
| Mean stage |  | -0.148 | -0.056 | -0.130 |
| Mean embryo dry weight |  |  | -0.285 | 0.043 |
| Number of embryos |  |  |  | **0.941*** |

## TABLE 4

**Digestive tract contents of 27 gravid Cayman gambusia collected from Little Salt Creek on 25 January 1997. Number of digestive tracts with prey frequency ranges are given.**

| Prey Type | Prey Frequency Range | | | | | |
|---|---|---|---|---|---|---|
| | 1 | 2–4 | 5–10 | 10–100 | 100–250 | 250–500 |
| Foraminiferida | 2 | | 1 | 5 | 5 | 2 |
| Gastropoda spp. | 1 | 1 | | | | |
| Polychaeta spp. | 3 | | | 1 | | |
| Araneae | 2 | | | | | |
| Prostigmata | 4 | | | | | |
| Oribatida | 3 | | | | | |
| Pseudoscorpiones | 2 | | | | | |
| "Planktonic" Ostracoda | 4 | 2 | | | | |
| "Harpactacoid" Copepoda | 2 | | 1 | 1 | | |
| Amphipoda | 2 | | | | | |
| Decapoda | 1 | | | | | |
| Isopoda | | 2 | | | | |
| Stomatopoda (larvae) | 2 | | | | | |
| Tanaidacea | 2 | | | | | |
| Coleoptera | 3 | 1 | | | | |
| Diptera | 3 | 11 | 1 | | | |
| Hemiptera | 6 | 2 | | | | |
| Hymenoptera | 7 | 9 | 3 | | | |
| Orthoptera | 2 | | | | | |
| Thysanoptera | | 1 | | | | |
| Insecta | 9 | | | | | |
| Osteichthyes | 2 | 1 | | | | |
| Pollen grain | 1 | | | | | |
| Seed pod | 6 | | | | | |

species of digenean parasites, too immature to be identified, were observed in the intestines of Cayman gambusia.

## Status

Cayman gambusia is a distinctive poeciliid species that appears to be restricted to the central mangrove area of North Sound, Grand Cayman. Collections made over the course of 4 extensive surveys of ponds throughout the Cayman islands failed to produce this species from any other areas. This unique species of *Gambusia* is of special concern in light of its low prevalence and pressures imposed upon the mangroves of Grand Cayman, by both the threat of development as well as large catastrophic tropical storms that frequent this region.

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# Gulf and Caribbean Research

First Report of the Mayan Cichlid, *Cichlasoma urophthalmus* (Günther 1862) Collected in the Southern Littoral Zone of Lake Okeechobee, Florida

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## SHORT COMMUNICATION

# FIRST REPORT OF THE MAYAN CICHLID, *CICHLASOMA UROPHTHALMUS* (GÜNTHER 1862) COLLECTED IN THE SOUTHERN LITTORAL ZONE OF LAKE OKEECHOBEE, FLORIDA

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## INTRODUCTION

The Mayan cichlid, *Cichlasoma urophthalmus* (Günther 1862), is a native freshwater fish of tropical America. Its natural distribution ranges from southern Veracruz in Mexico to Nicaragua, including the entire Yucatan peninsula (Miller 1966, Kullander 2003). Mayan cichlids in their native region have been found in both fresh and brackish waters (Miller 1966). Several studies have documented their ability to thrive in a wide range of salinities (Martinez-Palacios et al. 1990, Chavez-Lopez et al. 2005) and water temperatures (Stauffer and Boltz 1994). Mayan cichlids generally prefer well-oxygenated habitats with deep, transparent water and submerged aquatic vegetation (Chavez-Lopez et al. 2005), but also have been observed to withstand virtual anoxia for more than two hours in the laboratory (Martinez-Palacios and Ross 1986). Their robustness has allowed this species to thrive in many different aquatic habitats in the south Florida region including Florida Bay (Loftus 1987), Naples on the west coast (Faunce et al. 2002), Lake Osbourne in West Palm Beach (Fuller et al. 1999), and as far north as Charlotte Harbor (A. Adams, pers. comm., Mote Marine Laboratory, Sarasota, FL). It remains unclear how this species was initially introduced to Florida, but it is speculated that it was accidentally released from private aquariums. Loftus (1987) and Trexler et al. (2000) note that although initial densities of Mayan cichlids in Florida Bay fluctuated when first introduced, they have since reached higher numbers and have become established in that region.

Here we report the collection of five juvenile Mayan cichlids in Lake Okeechobee, a large, shallow subtropical lake (26°60'N, 80°50'W) (Figure 1). These specimens were collected on 4 and 10 November 2003 in the southern littoral zone of Lake Okeechobee at a site off the southwest tip of Torry Island (26°42'N, 80°44'W). The lake stage of this densely vegetated area is highly variable. Nico (in press) reported a single collection of 16 juvenile Mayan cichlids in 2001 from a backwater area in the rim-canal along the NE portion of Lake Okeechobee (Figure1); an area that is hydrologically isolated from the lake proper, except for several navigation locks and water control structures.

## MATERIALS AND METHODS

Five specimens were collected with a 1 m$^2$ aluminum throw trap during fish surveys by the South Florida Water Management District (SFWMD). After deployment of the trap, vegetation was harvested using rakes, aquatic weed cutters, and/or manual removal of plants. Fish were collected with a 600 µm mesh "D-shaped" dip net. Net sweeps were repeated until six consecutive sweeps yielded zero fish. Fish were placed in plastic Whirl-Pak™ bags and kept on ice. Standard lengths (SL, mm, ± 0.01) were measured using a digital micrometer and specimens were weighed (wet weight, ± 0.01g). Fish were then fixed in 10% formalin and then transferred to 70% ethanol. One specimen each was submitted to the Florida Museum of Natural History, Gainesville, Florida, and to the Ichthyology Museum at the University of Southern Mississippi. To assess diet, we removed and examined the digestive tracts of two additional specimens. Gut contents and prey items were identified to lowest possible taxonomic category.

## RESULTS

*Cichlasoma urophthalmus* specimens were collected and identified based on the following combination of characteristics: shape of body and snout, dark bands on body side, intense dark blotch at the caudal fin base, dorsal spine count of XV (range from XV–XVII), anal fin spine count of V (range is V–VI), and conical shape of teeth (identification verified by Nico).

The range of fish ($n = 5$) collected was 18.71 to 34.94 mm ($\bar{x} = 25.07$ mm SL) and wet weight ranged from 0.16 to 1.17 g ($\bar{x} = 0.51$ g). Four of the five Mayan cichlids were collected from dense beds of *Hydrilla verticillata*, an exotic submerged aquatic plant (Havens 2003). The remaining individual was captured in a giant bulrush stand (*Scirpus*
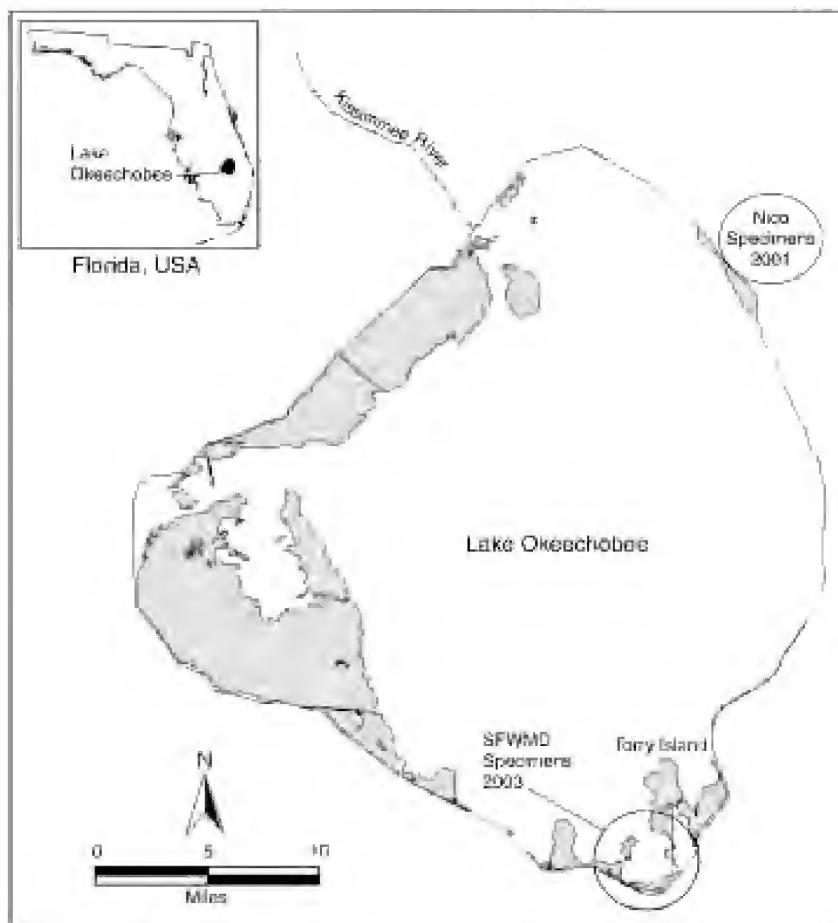
Figure 1: Map of Lake Okeechobee, Florida, noting collection sites of this study and those of Nico 2001.

*californicus*). Other fish collected included the golden top-minnow (*Fundulus chrysotus*), bluefin killifish (*Lucania goodei*), flagfish (*Jordanella floridae*), eastern mosquitofish (*Gambusia holbrooki*), least killifish (*Heterandria formosa*), and sailfin molly (*Poecilia latipinna*).

Digestive tract analysis of two specimens (18.71 and 22.32 mm SL) yielded more than 99% detritus, fish scales the posterior half of an unidentified amphipod, and partially digested remains of a terrestrial insect.

## Discussion

Although the mode of introduction of Mayan cichlids in Florida cannot be determined conclusively, it is evident this species is capable of expanding its range. Mayan cichlids were reported in southern Florida for the first time in 1983 from samples collected in Everglades National Park (Loftus 1987). It was determined from that study that Mayan cichlids were restricted to the Taylor Slough drainage basin; however, Shafland (1996) reported that Mayan cichlids had spread and become abundant in several canals and rivers of southern Florida, including the C-

111, and the more northern C-7 canal. Faunce et al. (2002) have also reported Mayan cichlids in Naples, on Florida's west coast, and further north and east in West Palm Beach.

Opportunistic feeding behavior and a tolerance to a wide range of salinities are characteristics that may facilitate the success of the Mayan cichlid in new habitats and enhance its spread into new locations in Florida. Arthington and Mitchel (1986) have suggested that this species is a generalist feeder. Bergman and Motta (in press) support this claim with fish collected from Big Cypress National Preserve concluding that the Mayan cichlid demonstrates a generalist diet throughout its ontogeny and primarily consumes detritus, vegetation, gastropods, crustaceans, insects, and fish. Caso-Chavez et al. (1986) and Chavez-Lopez et al. (2005) found that Mayan cichlids ate mostly plant material and supplemented their diet with crustaceans, insects, and mollusks. In contrast, in Mexico, the Mayan cichlid has been classified as a carnivore because it appears to prey primarily on small animals (Martinez-Palacios and Ross 1988).

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The distribution and feeding ecology of the Mayan cichlid in Lake Okeechobee is largely unknown, although our limited gastro-intestinal tract analysis would seem to support generalist feeding behavior, at least for juveniles. The systematic collection of juvenile Mayan cichlids in the southern end of Lake Okeechobee, Nico's (in press) collection from the NE rim-canal, and anecdotal accounts of captures by fishermen from widely distributed locations throughout Lake Okeechobee suggest this species is ubiquitous in the system. However, we have yet to observe any recruitment, spawning, or nesting behavior; key factors that define an established community (Loftus 1987). An adult Mayan cichlid was recently captured in the Kissimmee River, a major tributary that flows into the northwestern corner of the lake (L. Glenn, pers. comm., SFWMD, West Palm Beach, FL). To our knowledge, this is the most northern specimen collected in the Lake Okeechobee watershed.

Further research on the potential impact of Mayan cichlids on native fish populations and its role in the trophic structure of Lake Okeechobee is warranted since the species appears to have the potential to become a common member of the littoral zone community. The distribution of invasive fish species such as the Mayan cichlid in south Florida is highly variable and is possibly a function of habitat preference, spatial relationship of sampling area to point of introduction, and ambient temperature changes (Trexler et al. 2000). For Mayan cichlids in the Everglades, Trexler et al. (2000) reported that annual minimum temperature affected species abundance, and that introduced species required time to expand from their point of introduction. Further investigation may provide insight into how and where the Mayan cichlid was introduced to Lake Okeechobee, and how to monitor and regulate its population growth.

## ACKNOWLEDGMENTS

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# Gulf and Caribbean Research

Recent Observations of the Whale Shark (*Rhincodon typus*) in the Northcentral Gulf of Mexico

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## SHORT COMMUNICATION

# RECENT OBSERVATIONS OF THE WHALE SHARK (*RHINCODON TYPUS*) IN THE NORTHCENTRAL GULF OF MEXICO

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### INTRODUCTION

The whale shark (*Rhincodon typus* Smith, 1828) is the world's largest fish, reaching 15 meters (m) and 18 metric tons (Colman 1997) and is found in all tropical and warm temperate seas (Compagno 2001). The whale shark is listed as 'vulnerable' by the International Union for the Conservation of Nature and Natural Resources (IUCN 2004) and is included in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 2004). Little is known about whale sharks in the Gulf of Mexico (Gulf). Only reports on the occurrence of whale sharks off Texas (Baughman 1950, Baughman and Springer 1950, and Hoffman 1981) and accounts of the occurrence and feeding in the northcentral Gulf (Gudger 1939, Springer 1957) are available. Due to the lack of information on whale sharks in the Gulf, we developed a survey (http://www.usm.edu/gcrl/whale-shark_survey) to compile records of recent sightings and associated observations that are summarized here.

### MATERIALS AND METHODS

Details of whale shark sightings were obtained from recreational fishers, charter fishing operators, and divers by personal interview or the internet survey. Information requested from individuals who sighted whale sharks included date, location (e.g., GPS coordinates, direction and distance from a coastal landmark, or identifier number of specific offshore petroleum platforms), estimated total length (TL, ft), number of individuals, behavior (e.g., swimming, feeding), and associated fishes. Sightings greater than two years old were not included in the database unless sufficient documentation (e.g., log entry) of their validity was provided.

### RESULTS

Interviews provided information for 26 sightings involving 46 whale sharks between July 2002 and November 2004. Additionally, four large aggregations (30–100 individuals) of whale sharks were also reported; however, information reported for the aggregations was scant and not included in the data analysis. Nineteen sightings were of individual whale sharks, with seven sightings consisting of two to seven sharks. The seasonal distribution and the number of whale sharks are shown in Figure 1. Sightings occurred in waters with depths from 20 to 1,000 m. Most whale sharks (80%) were observed swimming horizontally near the surface of the water, while the others were observed in vertical profile, apparently suction-feeding on small prey.

All observations occurred between May and November with 83% of the sightings occurring between July and October (Figure 2). There was a prevalence of sightings southwest of the Mississippi River Delta during summer and northeast of the Delta during the fall (Figure 1). Many sightings (63%) occurred at or near petroleum platforms (Figure 3). Estimated size (feet converted to meters) ranged from 3.7 to 10.7 m TL (*n* = 41, Figure 4). Gender was not noted.

Ten teleost and two shark species were observed with whale sharks during 69% of the sightings (Table 1). Tunas were the most commonly reported, particularly blackfin, *Thunnus atlanticus*, skipjack, *Katsuwanus pelamis*, and yellowfin, *Thunnus albacares*. The authors and colleagues caught blackfin and skipjack tuna from large schools of tuna associated with two whale sharks in September 2002.

### DISCUSSION

The information reported here represents recent accounts of whale shark sightings in the northcentral Gulf. These data plus unpublished accounts (K. Mullins, NOAA Fisheries, Pascagoula Facility, per. comm.) suggest that whale sharks occur frequently in the northern Gulf during warmer months, entering the northcentral Gulf from the west or southwest in the late spring/early summer. They appear to move northeastward during the fall, and are perhaps absent during the winter. These apparent seasonal patterns are based only on surface observations. We assume there is no seasonal variability in vertical position of this species within the Gulf and thus no bias in these observations. Although whale sharks are considered to be
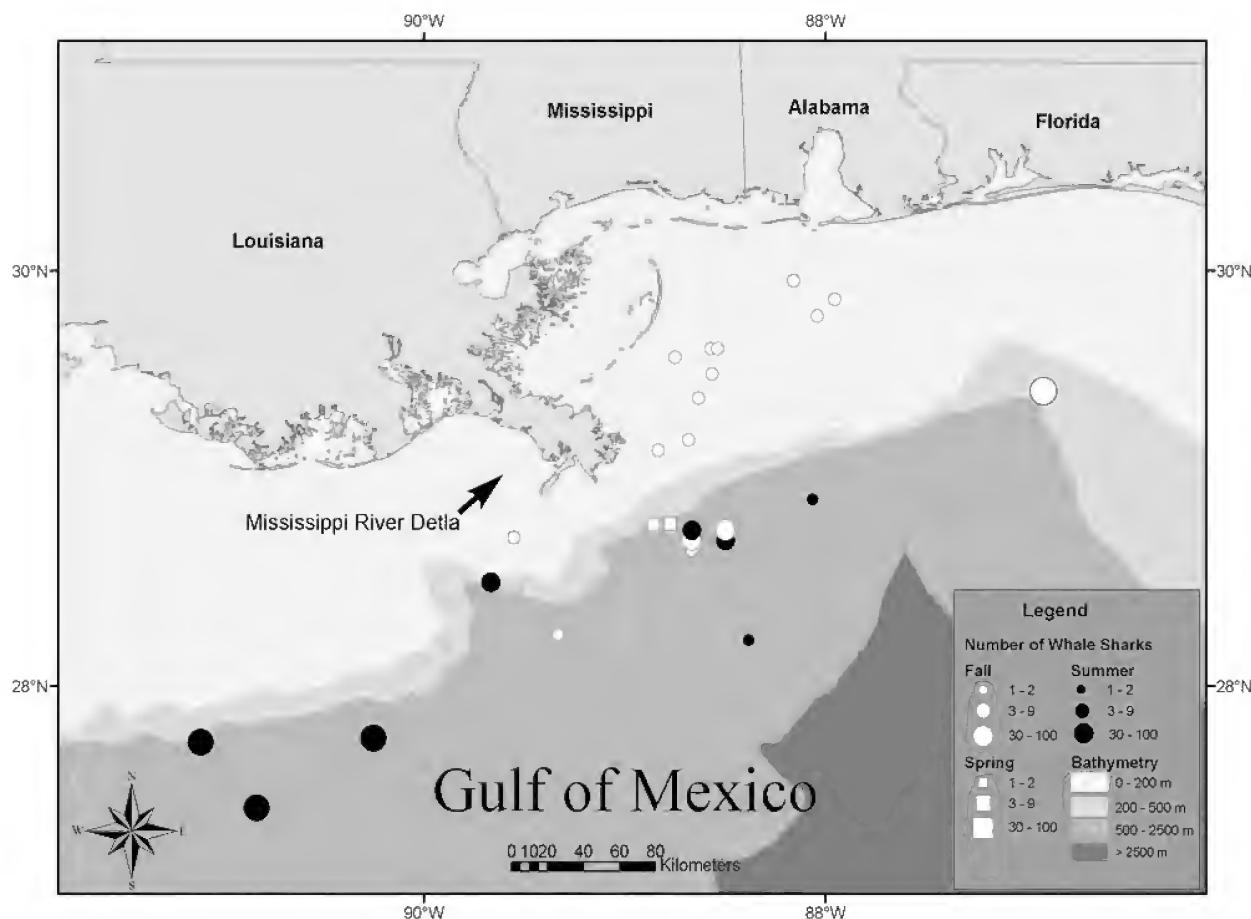
**Figure 1. Locations of whale shark, *Rhincodon typus*, sightings in the northcentral Gulf of Mexico between 2002 and 2004. Some symbols may overlap due to the proximity of individual sightings. Only the smallest symbol for spring is shown.**

highly migratory throughout much of their range (Eckert and Stewart 2001), available data provide no insight into whether whale sharks in the northern Gulf are transient or comprise a resident population.

Aggregations of whale sharks (up to 30 individuals) were previously reported in the northern Gulf (Gudger 1939; W. Driggers, NOAA Fisheries, Pascagoula Facility, per. comm.). The significance of whale shark aggregations is unknown, but Colman (1997) reported that aggregations may occur in areas with dense prey. The four large aggregations as well as the majority of other sightings reported here occurred at or near petroleum platforms, which function as fish aggregating devices (Franks 2000). However, the predominance of sightings at petroleum platforms is likely attributable to the use of platforms as preferred recreational fishing destinations. Whale sharks themselves attract other fishes (Gudger 1941, Baughman and Springer 1950, Hoffman et al. 1981, Clark and Nelson 1997), and we report the highest diversity of pelagic fishes documented in association with whale sharks.

Although most whale sharks in this study were observed swimming horizontally, it was not always evident that they were feeding. However, 20% of the whale sharks were observed suction-feeding while in vertical profile, similar to reports by Gudger (1941), Springer (1957) and Hoffman et al. (1981). Springer (1957) reported five whale sharks feeding vertically on small fishes in a school of blackfin tuna in the northcentral Gulf. Running-ripe male and female blackfin tunas caught during the author's 2002 whale shark encounter regurgitated small clupeids on deck. However, we could not determine if the whale sharks were feeding on clupeids or the spawn of the tuna. Colman (1997) suggested that whale sharks and associated fishes may feed on the same prey, and Heyman et al. (2001) reported whale sharks feeding on snapper spawn, suggesting that this feeding behavior may also be occurring here.

Seasonal distribution of whale sharks in the northcentral Gulf may be influenced by hydrologic/oceanographic features (e.g., Loop Current, Mississippi River plume, convergent zones, upwellings, temperature discontinuities).
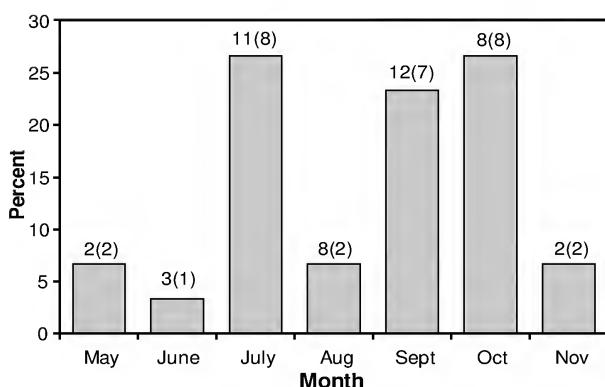
Figure 2. Percent occurrence by month of whale sharks, *Rhincodon typus*, observed in the northcentral Gulf of Mexico from July 2002 to November 2004. Numbers above histograms indicate sample size, and numbers in parentheses indicate the number of sightings.



Figure 4. Length frequency plot of whale sharks, *Rhincodon typus*, observed in the northcentral Gulf of Mexico from July 2002 to November 2004. Numbers above histograms indicate sample size.



Figure 3. An estimated 8 m whale shark, *Rhincodon typus*, near a sport fishing boat at a petroleum platform in the northcentral Gulf of Mexico on October 30, 2003.

Such features provide optimal conditions for the production of plankton (Govoni et al. 1989, Richards et al. 1993), a food source of whale sharks (Colman 1997). These features also aggregate primary consumers such as crustaceans, small fishes, and jellyfish which are also known prey of whale sharks (Gudger 1941, Colman 1997, Heyman et al. 2001), thereby creating spatially discrete feeding areas. Finally, Wilson et al. (2001) noted that whale sharks may time their seasonal movements to coincide with localized productivity events or behavioral changes in their prey.

The individuals reported here ranged from 3.7 to 10.7 m TL and 56% appear to be immature as Joung et al. (1996), Beckley et al. (1997) and Wintner (2000) reported

that maturity occurs at 9.0 m TL and 10.5 m TL for males and females, respectively. These data, along with Baughman's (1955) reported collection of an aborted whale shark egg case off Texas, imply that the Gulf may be a whale shark nursery area, as was suggested by Gudger (1939).

Limited data are available on life history, movement, and habitat requirements of whale sharks in the northcentral Gulf. Furthermore, their designation as "vulnerable" by IUCN and their listing by CITES demonstrate the need for greater understanding of this species throughout its range. Hoffmayer et al. (in press) proposed a plan of research, which includes population surveys, biological assessments, and habitat use evaluation to advance the scientific understanding of whale sharks in the Gulf for the develop of future management plans and protection measures for the species.

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# Introduction to Special Section on Research Activities at the Iztacala Campus of the Universidad Nacional Autónoma de México, México

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# INTRODUCTION TO SPECIAL SECTION ON RESEARCH ACTIVITIES AT THE IZTACALA CAMPUS OF THE UNIVERSIDAD NACIONAL AUTÓNOMA DE MÉXICO, MÉXICO

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In fall 2002, we had the opportunity to initiate a long-term collaboration with colleagues from the Iztacala campus of the Universidad Autónoma de México (UNAM), whose research interests focus on coastal and nearshore fishes and decapod crustaceans of Veracruz state, Mexico. This is an undergraduate campus with a strong Biology department whose faculty also maintain research programs despite limited sources of funding and a heavy teaching load. During the course of our initial visit, we realized that there is a wealth of unpublished undergraduate student research on understudied fish species. We developed a collaborative partnership in November 2002 between the faculty advisors from the Biology department at UNAM-Iztacala, the Department of Coastal Sciences, The University of Southern Mississippi and non-profit funding organizations interested in research in the Gulf of Mexico and Caribbean Sea. We selected exceptional senior and Masters theses from the Biology department at UNAM-Iztacala which were processed and submitted for peer-review in *Gulf and Caribbean Research* (see Peterson and Brown-Peterson 2004). The following publications are a continuation of the partnership.

## ACKNOWLEDGMENTS

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